DOCTOR OF PHILOSOPHY

Sustainable development
Why is it not delivering on its promises?

Gonzalez Redin, Julen

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Sustainable development:

Why is it not delivering on its promises?

Julen Gonzalez Redin

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Eskerrik asko eta irakurketaz gozatu!
¡Muchas gracias y que disfruten de la lectura!
Thank you very much and enjoy reading!

Declaration

I declare that this thesis, presented for the degree of Doctor of Philosophy (Ph.D.), has been composed solely by myself and that I have consulted all the references cited in it. The thesis is a record of work that I have done and it has not been submitted, in whole or in part, in any previous application for a degree.

Appendix A shows a detailed description of the different research outputs delivered during the PhD period. Co-authorship of Prof. Iain J. Gordon, Prof. Terence P. Dawson, Dr. J. Gary Polhill, Dr. Rosemary Hill and Dr. Mark Cutler represents their input to this thesis as supervisors, in the form of comments, suggestions, advice, and discussions of all aspects of the research.

………………………………………………………………… Julen Gonzalez Redin

I confirm that the conditions of the relevant Ordinance and Regulations have been fulfilled in relation to this thesis.

……………………………………………………………….… Dr. Mark Cutler

Abstract

At the Rio Conference in 1992, the sustainable development agenda promised a new era for natural resource management, where the wellbeing of human society would be enhanced through a more sustainable use of natural resources. Several decades on, economic growth continues unabated at the expense of natural capital – as evidenced by natural resource depletion, biodiversity loss, climate change and further environmental issues. Why is this happening and what can be done about it?

This research examines what socio-economic and governance factors affect sustainability in complex coupled social-ecological systems. Furthermore, it analyses the role of power relations and imbalances between economic and conservation forces with regard to sustainable development. The original contribution to knowledge of this thesis is based on one conceptual and two empirical (Agent-Based) models. These explore, through several case-studies, the potential of different future scenarios in fostering synergies and win-win contexts of ecosystem services and socio-economic indicators.

Overall, the research showed the complex and interconnected relationship between the economy and natural systems, and between economic and conservation forces, in coupled social-ecological systems. Addressing complex sustainability issues requires the use of integrative, holistic and interdisciplinary approaches, in addition to considering the particular socio-economic, cultural, political and environmental contexts of the social-ecological system being analysed. The models demonstrated that the current economic system requires an ever-increasing use of natural resources, and that the economy does not protect the natural capital on which it depends. This is based on a disjunction of the economic and conservation elements upon which the sustainable development paradigm is founded. Furthermore, several socio-economic and governance factors appeared to be key for diminishing sustainability in coupled social-ecological systems; namely, the type of economic and production systems, the particular use of monetary debt, technological development, and weak conservation forces (both top-down and bottom-up). However, results also showed alternative scenarios where these same factors could be redirected to enhance social-ecological sustainability. This dual role supports the argument that the current economic system is not inherently (i.e. by definition, *per se*) unsustainable. Rather, the specific use of economic mechanisms and behaviour of economic entities, as well as their decisions and relationships with the environment, show a tendency to increase unsustainability. Hence, short- and medium-term sustainability can be enhanced by developing mechanisms that start shifting capitalist forces to support environmental conservation; here, the role of Payments for Ecosystem Services will be essential. Enhancing long-term sustainability, however, may require a further paradigm change – where economic and production systems integrate, and fully account for, externalities and the value of natural capital, thus human society is embedded within the wider, and more important, natural environmental system.

Chapter 1: Introduction

"*The earth, the air, the land and the water are not an inheritance from our forefathers but on loan from our children. So we have to handover to them at least as it was handed over to us*".

– Gandhi (Indian activist, as cited in Kaushik, 2010, p.1).

1.1 Sustainable development

It is widely recognized that sustainability represents the greatest challenge for humanity in the Anthropocene (Wu, 2013). A large number of words have been written on the complex set of environmental problems facing humanity, such as climate change, biodiversity loss, natural resource depletion, especially as compared to the number devoted to serious solutions(Costanza, 2007; UN, 2016). The debate about the role economic growth plays, concerning these problems, has been rapidly gaining importance over the last decades. Basically, the capitalist economic system is not embedded within the wider, more important natural environmental system (Berkes and Folke, 1998). This is because the current economic paradigm endures under the growth strategy initiated by the Bank of England around 1700 (Martenson, 2010), where the economic system is not constrained by the biophysical limits within which natural resource systems operate. The result is a strong positive relationship between income per capita and demand for natural resources, which disconnects the economic system from natural capital (Ward *et al*., 2016). As a consequence, the future availability of natural resources, e.g. food, water energy, minerals, as well as human wellbeing, is critically endangered (Costanza *et al*., 1997; World Economic Forum, 2014).

There is an obvious need for a paradigm shift if natural resource consumption is to be decreased while the needs of the growing human population are met. If economic growth is not absolutely decoupled from environmental pressures, the systems that support life on this planet are going to collapse in the near future (Smith *et al*., 2010). As a result, most current societies have been increasingly concerned about the sustainable development of their economies since the world-wide oil crisis of the mid 1970s (Schafer, 2014). Since then, different pathways towards a more sustainable economy have been proposed, including steady state and degrowth approaches (Daly, 1991; Jackson, 2009), green growth (OECD, 2011), circular economy (Pearce and Turner, 1990), among others. Yet, none of these have been truly successful at enhancing a more sustainable economic system (Smith *et al*., 2010). One of the problems lies in the fact that some existing policies are based on the science of the 1950s, '60s and '70s (largely disciplinary), therefore, they are not designed to address the current problems in natural resource management (De Greene, 1993; Gunderson *et al*., 1995; Lee, 1993; Meadows and Robinson; 1985). Back then, issues with regard to natural resources were considered to be largely local, reversible, and direct; today we know that impacts are changing rapidly, potentially irreversibly, and occur geographically (Daily, 2000) and economically (Lambin *et al*., 2001), at a global scale. Moreover, past scientific approaches were based on mono-disciplinary ideas that neglected system complexity (Gleick, 2003; Holling and Meffe, 1996; Ludwig, 2001; Pahl-Wostl, 1995), while today it is widely recognized that unsustainable development cannot be attributed to a single cause, but rather to a set of multivariate, non-linear, cross-scale and dynamic factors (Holling *et al*., 1998).After all, unsustainable development could be rooted to human failure with regard to understanding the links between social, ecological, and economic systems. Thus, there is a need for further systemic, holistic, integrative and interdisciplinary approaches that allow better understanding of the interrelation between the economy and the environment (Binder *et al*., 2013).

In this regard, a growing body of literature is treating social and ecological systems as a single coupled and dynamically complex system (Folke, 2006; Gunderson and Pritchard, 2002; Ostrom, 2007, 2009); these systems are composed of people and nature, and defined as social-ecological systems (SES) (Redman *et al*., 2004). SES science is attracting interdisciplinary approaches that explore which combinations of factors lead to (un)sustainable and unproductive SES. This issue was highlighted in The World Economic Forum (2014), arguing that the state of natural resources and their distribution was increasingly being threatened by various drivers and pressures. In this regard, a number of authors have studied the extent to which different combination of socio-economic, political, cultural, environmental, and other variables could be leading to the unsustainable use of natural resources in complex SES, thereby increasing resource collapses and high costs for humanity (Ostrom, 2007). However, the multiple timescales of ecological change, and the complex features of the social and economic dimensions, make the analysis and interpretation of these variables a difficult and challenging task (Brock and Carpenter, 2007). Thus, there is a need to develop interdisciplinary, integrative empirical models of SES that help provide answers to what combinations of factors hinder sustainable development in such complex systems. Due to the wide-ranging nature of exploring sustainability in complex coupled SES, and to the multifaceted and abstract character of the research itself, an overwhelming number of situations and contexts come into play. Therefore, specific research questions are necessary, as well as a clear and contextualized definition of sustainable development, and what is referred to as SES. This research builds upon conceptual, empirical and spatially-explicit computer models to address the research aim and objectives posed below (section 1.2), with a special focus on interconnecting social-ecological systems (SES) and sustainability.

1.2 Research aim and objectives

The multiple elements of the PhD research project all aim to contribute to one broad and straight-forward, yet complex and challenging, central research question:

> *What hinders sustainable development under the current capitalist economic system, and is there a built-in bias towards environmental unsustainability?*

In order to answer this question, the following three specific research objectives will be undertaken:

1. To study what combinations of socio-economic and governance factors drive SES (un)sustainability in complex SES.

2. To investigate the relationship between (monetary) debt and SES (un)sustainability; specifically to study impacts exerted by debt-driven speculation and technological development and efficiency processes on SES (un)sustainability.

3. To examine the effect of economic and conservation powers (forces), and the conflicts and power (im)balances between them, on SES (un)sustainability.

These three research objectives form the basis of the research and methodological approach. The following section now outlines the research strategy and methodological approach of the thesis.

1.3 Contextualizing (social-ecological) sustainability in this thesis

The wide-ranging and multi-faceted nature of this research requires a clear contextualization of the sustainability with regards to SES. Hence, the objective is to apply the concept of sustainability as effectively and simply as possible, while respecting the nature and definition of the term. Sustainability is known for its three 'pillars', or the 'triple bottom line' (Holdren, 2008), where sustainability is achieved if economic development (economic pillar), social development (social pillar) and environmental protection (environmental pillar) are enhanced in a balanced way. At the same time, sustainable development, as defined in 1987 in the report called 'Our Common Future', Brundtland Commission (Brundtland Commission, 1987), refers to the "*Development that meets the needs of the present without compromising the ability of future generations to meet their own needs."*.

By combining both approaches, a sustainable SES can be considered one that is driven by sustainable development, where the latter refers to a socially inclusive and environmentally sustainable economic growth (Sachs, 2015). In particular, this thesis focuses on the economic and environmental dimensions of sustainability, that is, on exploring to what extent an environmentally sustainable economic growth is possible under the current system of economic growth. The reason for selecting the economic-environmental intersection is twofold: first, the concept of 'decoupling', which is a main interest in this thesis (analysed in Chapter 2), is principally focused on the disjunction between economic growth and environmental pressures (Smith *et al*., 2010). According to the OECD (2002), the term 'decoupling' refers to breaking the link between the growth in environmental pressure associated with creating economic goods and services. Thus, although the social dimension (e.g. poverty, wellbeing, inequality) will always be indirectly affected due to the interconnectedness of the three pillars of sustainability, this thesis focuses on the economic and environmental sustainability of SES (hereafter called '*SES sustainability*'). Second, focusing on the economic and environmental pillars allows the use of profit- and ecology-based indicators (e.g. monetary capital, natural resource stocks) to quantitatively track decoupling scenarios; these are normally easier to analyse, and obtain data from, compared to the generally more subjective social development indicators, e.g. well-being, social inequality.

Due to the SES focus of this research, sustainability (i.e. economic-environmental sustainability) needs to be applied under a SES perspective. In this regard, the United Nations General Assembly proposed, in 2015, 17 interrelated Sustainable Development Goals (SDGs) under the 2030 Agenda for Sustainable Development (UN, 2016). The SDGs cover all sectors of society and all aspects of sustainability, including poverty, hunger, health, education, climate change, gender equality, water, sanitation, energy, environment and justice. Interestingly, the SDGs report provides one SES-based approach focused on integrating ecosystem services (ES) into strategies for enhancing economic growth while protecting the environment (Wood *et al*., 2017). ES are the benefits that humankind obtains from nature directly and indirectly, usually categorised into provisioning, regulating, supporting and cultural services (MEA, 2005). The SDG report argues that there is a need to create policies and strategies that enhance synergies of ES in order to support environmentally sustainable economic growth (UN, 2016). Thus, this thesis will use this ES-based approach to focus on a key nexus for achieving economic and environmental sustainability highlighted in the SDG report: the relationship between climate change (SDG 13[1]: 'Climate Action'), food production (SDG 2[2]: 'Zero Hunger') and biodiversity conservation (SDG 15[3]: 'Life On Land') (see

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[1] "Take urgent action to combat climate change and its impacts".

[2] "End hunger, achieve food security and improved nutrition and promote sustainable agriculture".

[3] "Sustainably manage forests, combat desertification, halt & reverse land degradation, halt biodiversity loss".

CBD, FAO, UN Environment, UNDP, 2016)[4]. Because the selected SDGs are underpinned by the delivery of one or more ES (Wood *et al*., 2017), it is necessary to first understand the relationship between the three SDGs, as well as which ES could help achieving the selected SDGs. First, biodiversity conservation (i.e. SDG 15) is established as a key process for the achievement of food security (SDG 2), since all food systems depend on biodiversity to support productivity, soil fertility, water quality, and other ES (Gordon, Squires and Prins, 2016). At the same time, one of the biggest threats to biodiversity is habitat loss resulting from land clearing for pastoral and/or agricultural activities related to food production (MEA, 2005). On the other hand, the SDG report highlights climate change (SDG 13) as one of the main drivers of biodiversity loss (SDG 15), as well as the importance of conserving biodiversity to help reducing the risks and damages associated with negative impacts of climate change. Finally, the food production-climate change relationship is discussed in the SDG report around the potential impacts that climate change has on food production (Porter *et al*., 2014), as well as the importance of sustainable food and agricultural systems to help mitigate and adapt to climate change, such as organic agriculture (Muller *et al*., 2017).

In short, the SDG report highlights the interconnectedness of these three aspects of sustainability (biodiversity conservation, food production, climate change mitigation) and the need to enhance win-win-win strategies to achieve an environmentally sustainable economic growth. Based on this rationale, this thesis uses the nexus between climate change mitigation–food production–biodiversity conservation as a key driver of SES (un)sustainability. Thus, a sustainable scenario in this thesis, or *'SES sustainability',* is referred to one showing *win-win-win outcomes among biodiversity and those ES related to climate change and food production*. In particular, different specific ES indicators are considered for each of the three sustainability elements (i.e. SDGs). These are: carbon sequestration (Chapter 5) and reduction on carbon emissions (Chapter 4) regarding climate change mitigation; crude palm oil (Chapter 4) and sugarcane production (Chapter 5) regarding food production; and biodiversity

[4] Similar to the previously mentioned social pillar, the interrelated nature of the SDGs will also make this thesis to (indirectly) address other socially-oriented SDGs. However, the latter will not be analysed and discussed throughout the thesis.

conservation – e.g. the number of plant species (Chapters 4 and 5)[5]. Sections 1.5 and 1.6 in this chapter provide a detailed description of the research strategy, structure and methodological approach of each chapter.

The next section discusses the importance of addressing the climate change mitigation–food production–biodiversity conservation nexus in the specific type case-study areas selected.

1.4 Research context and case-study areas

The research adopts both conceptual (Chapter 3) and empirical case-study (Chapters 4 and 5) approaches (see section 1.6). This section analyses the rationale behind the particular type of case-study areas selected to explore (un)sustainability, as previously defined, in complex SES. Thus, these case-studies will be used to address the specific research aim and objectives posed in section 1.2. Note that further detailed information on each individual case-study is included in the corresponding chapters.

This thesis selected tropical regions as complex coupled SES. Tropical areas are characterized for being coupled human-natural systems where the interactions between the human society and the environment are strong, complex and dynamic (Folke *et al.*, 2002; Redman *et al.*, 2004). Tropical SES are different from other SES because of the higher degree of risk and uncertainty associated with natural resources extraction, the dynamic nature of human resources, and often unclear tenure (Ferrol-Schulte *et al.*, 2015). In particular, most tropical regions are characterized for presenting a key trade-off for global sustainability, which represents the historic conflict between the economy and environment that is addressed in this thesis, i.e. the "development *versus* protection" dichotomy (Hartshorn, 1995). This conflict is based on economic forces – driving (environmentally unsustainable) growth through land clearing and deforestation

[5] Note that the model presented in Chapter 3 is a conceptual model, while the models presented in Chapters 4 and 5 are empirical (all described in sections 1.5 and 1.6). While the empirical models integrate the ES and biodiversity indicators selected, no specific ES and biodiversity indicators are simulated in Chapter 3. The latter chapter rather analyses the dynamics of a conceptual natural resource, where 'SES sustainability' is achieved through win-win results regarding natural resource stocks and other economic indicators.

– directly opposing environmental forces – driving land conservation through, for example, restoration and protection (Hill *et al*., 2015; Malhi *et al*., 2014). This conflicting scenario affects multiple SDGs, including those representing SES sustainability in this research, and provides a suitable context to address a key research objective of this thesis, based on addressing the current decoupling between economic growth and environmental pressures.

Furthermore, the interest on studying the SES sustainability of tropical SES is twofold:

(1) Tropical regions lie at the interchange of SES sustainability as defined in this thesis (see previous section), i.e. achieving win-win-win scenarios with regards to food production–climate change mitigation–biodiversity conservation. First, improving agricultural productivity in the tropics will be critical for feeding the growing human population (Fedoroff *et al*., 2010), where a 50% increase in food production will be needed by 2050 to sustain the rising food demand (Nellemann *et al*., 2009), as well as ending hunger, achieving food security, improving nutrition and promoting sustainable agricultural practices (Swamy *et al*., 2018). Second, there is a need to reduce emissions from tropical deforestation and degradation to halt global warming (Angelsen, 2008). Tropical forests, therefore, serve as an important medium for urgent action to combat climate change and its impacts (Swamy *et al*., 2018). Third, the sustainable use of tropical forests plays a key role in conserving terrestrial ecosystems, as well as halting and reversing biodiversity loss (Swamy *et al*., 2018). Land-use change (LUC), driven by the expansion and intensification of agriculture and plantations (Foley *et al*., 2005), is the main cause of biodiversity and ES loss in tropical regions, which are one of the biologically richest ecosystems in the world (Harrison *et al*., 2014; Molotoks *et al*., 2017). In short, tropical areas provide a research opportunity to study the above-noted ES and biodiversity trade-offs, which are a key aspect to achieve global sustainability as previously discussed (see UN, 2016).

(2) Although the SDGs are globally important and applicable to every country, they are especially relevant for tropical countries (Swamy *et al*., 2018). Developing countries are generally located in the tropics, which face most of the sustainability issues included in the SDG report. Thus, exploring pathways to achieve environmentally sustainable economic growth is especially relevant in these areas.

Among tropical areas, Indonesia and the Wet Tropics of Queensland, Australia, are selected as case-study areas for this thesis (see Figure 1.1). In particular, both areas are focal points for achieving global sustainability, and above all with regard to SES sustainability as defined in this thesis, i.e. food production–climate change mitigation–biodiversity conservation. In regard to Indonesia, this country has the highest plant species richness in the world (ICCT, 2016), while being the world's biggest producer of palm oil with an objective of near doubling the area for oil palm cultivation from 2015 to 2020 (UNDP, 2015). Furthermore, Indonesia is one of the world's top five greenhouse gas (GHG) emitting countries, above all from LUC (e.g. deforestation for palm oil production). Thus, the Government of Indonesia set the goal, in 2011, to reduce emissions by 2020 to 26% below 2011 values (Paltseva *et al.*, 2016). The extent to which the government will be able to achieve these three opposing goals for 2020 and further (Republic of Indonesia, 2016) is a relevant question regarding the conflicting SDGs and to achieve global sustainability. Similarly, the Wet Tropics of Australia is one of the most biologically diverse areas in the world, with forests embracing 35 international global biodiversity hotspots, and the only region in the world to include two adjacent World Heritage Areas–the Wet Tropics World Heritage Area and the Great Barrier Reef. At the same time, this biological richness is threatened by the expansion of sugarcane plantations, which is a key rural industry in Australia (AgriFutures, 2017). Thus, land clearing and deforestation is still a main cause of biodiversity loss and GHG emissions in the North-East of Queensland and Australia overall (Neldner *et al.*, 2017).

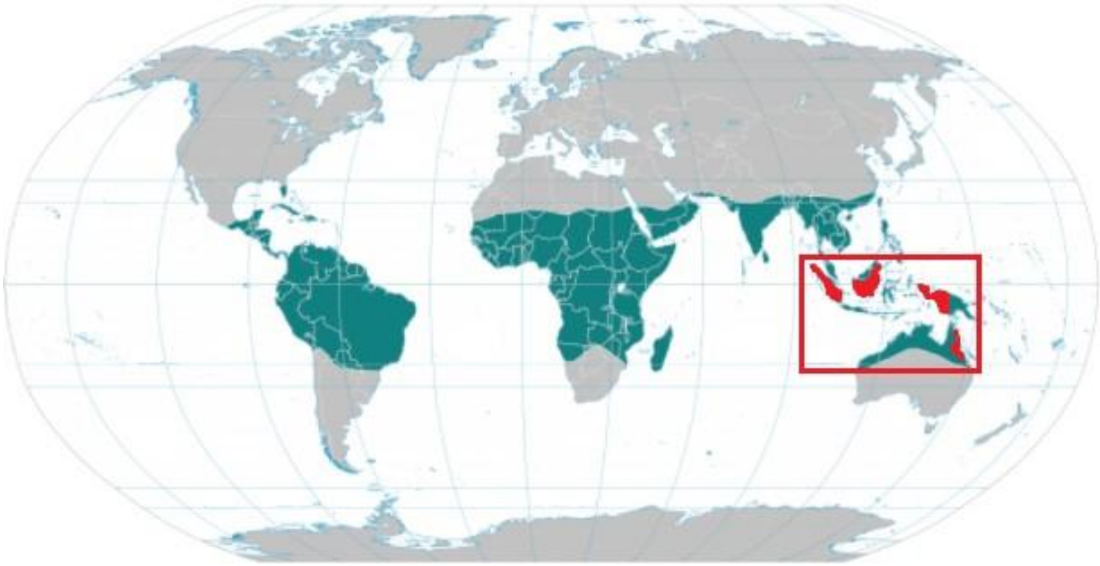

Figure 1.1: Global geographic location of tropical regions (dark green) and case-study areas (red).*Source*: author.

Both Indonesia and the Wet Tropics of Queensland present, therefore, a very similar trade-off regarding food production, biodiversity conservation and climate change mitigation. However, the main interest in selecting them as tropical SES case-studies lies on their different socio-economic, governance and political contexts. Thus, the same trade-off is being managed under two different scenarios – opposing in some aspects – that represent different socio-economic realities. This situation provides a research opportunity to study what specific socio-economic and governance factors in each country are key drivers of similar synergies and trade-offs among food production, climate change mitigation and biodiversity conservation – which is the first research objective of this thesis (see section 1.2).

One of the main differences between Indonesia and the Wet Tropics lies in those economic forces driving forest clearance for agricultural production. While, in Indonesia, deforestation forces are stronger than those driving forest protection –as is the case for most regions in tropical developing countries (Hill *et al*., 2015) – the opposite is the case in the Wet Tropics of Australia, where protected areas increased by around 20 percent from 1999 until 2015, with a total of 50 percent of land currently protected (DSITI, 2016). An obvious differentiating element between both regions is related to economics–where the governments of Queensland and Australia have more funding available for conservation compared to those from developing countries (e.g. Indonesia). This creates a context were wealthy developed countries can allocate more capital to environmental conservation and, to a certain degree, protect the environment from the rough edges of the market economy. Developed countries have achieved substantial economic growth and development, therefore, they are able to afford to focus on environmental goals because basic living necessities have been achieved (Omoju, 2014). This is not the case for developing countries, such as Indonesia, where halting environmental pressures may undermine economic growth and competitiveness, whose economies depend on natural resources (Omoju, 2014).

Besides this, there are other, less obvious, aspects that reinforce the presence of stronger economic forces driving agricultural production than protection in Indonesia, as compared to the Wet Tropics. A key economic factor in this regard is monetary debt – whose relationship with regards to environmental sustainability is analysed in this thesis (see research objectives, section 1.2). There is a high dependency of Indonesian palm oil companies on external funding through credit facilities from overseas banks (Forest and Finance, 2016). This additional capital is used to finance palm oil production through land clearing and deforestation, which in turn increases biodiversity loss and GHG emissions (Fitzherbert *et al*., 2008; Koh and Wilcove, 2008; Pearson *et al*., 2017). Therefore, this scenario provides a suitable context to study the debt-sustainability relationship in Indonesia (explained in section 1.6). The Wet Tropics, on the other hand, is characterized for having strongly institutionalized environmental conservation – including biodiversity and climate change mitigation –supported by multi-layered and committed conservation governance, as well as different social actors and entities (Hill *et al*., 2010, 2015bc). This is an atypical situation for a tropical region, considering that tropical areas, e.g. Indonesia, which are generally located in developing countries, are characterized for having weak governance, corruption, and other issues enhancing environmental unsustainability (Hill *et al*., 2015a; OECD, 2016). Regardless of the strong conservation force present in the Wet Tropics, this region – together with the rest of north-east of Queensland (Australia) – is still facing the previously described trade-offs between land clearing for food production, biodiversity conservation and climate change mitigation (Neldner, *et al*., 2017; Species Technical Committee and Laidlaw, 2017; Taylor, 2010), i.e. SES (un)sustainability in this thesis.

In short, the different contexts of Indonesia and the Wet Tropics of Queensland, Australia, combined with the presence of the same, or very similar, trade-off among ES and biodiversity in both regions, provides a research opportunity to contribute new insights with regards to what causes SES (un)sustainability in complex SES. The overall aim of the research in these chapters is, therefore, to examine what combination of factors may be hindering sustainable development under the current economic system.

1.5 Research strategy and methodological approach

The concept of SES sustainability and the case-study areas introduced above highlight the complexity of the area within which this research is positioned. This research recognises the growing debate within both policy and academic discourses for a more integrated, holistic, interdisciplinary and cross-scale approach to sustainability and sustainable development (Binder *et al*., 2013). At the same time, however, it recognises the need to link the generally broad and theoretical approaches to economic-environmental decoupling issues with more specific, empirical and spatially-explicit approaches, that use SES- and ES-based approaches to address ES trade-offs and bundles in complex SES. For this aim, novel methods and frameworks are needed that are able to balance, and integrate, theory and practice across different disciplines, as well as link top-down with bottom-up modelling perspectives. This section will outline the overall strategy and general methodology adopted for this research.

The overall PhD research strategy was divided in three separated stages: i) conceptual and theory-developing stage (Chapter 2); ii) exploratory quantitative modelling stage (Chapters 3-5); iii) results integration and discussion stage (Chapters 6-7). Figure 1.2 shows the links among the thesis research aims (for each chapter) and the particular methods and approaches used to address each of them.

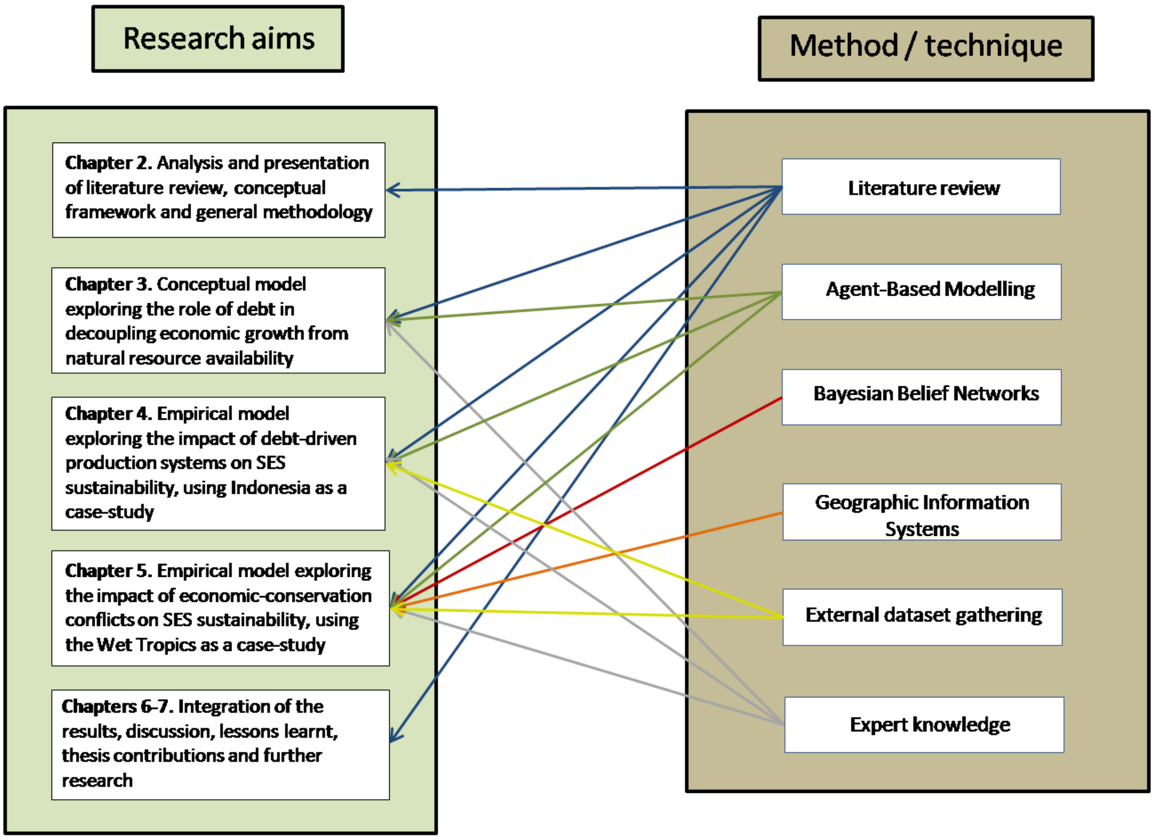

Figure 1.2: Research strategy and methodological approach of the thesis. Arrows represent the links between the specific method/technique used for within each chapter and research aim. *Source*: author.

The thesis begins with the review of existing literature from a number of disciplinary perspectives related to sustainable development (see Chapter 2). The thesis is interdisciplinary, thus incorporating elements from disciplines such as social-ecological systems science, ecological macroeconomics, and ecosystem services and conservation, which in turn have their foundations in (ecological) economics and sustainability science more generally. The literature review served as a theoretical foundation for selecting the research objectives (see previous section), as well as for building the conceptual framework presented in Chapter 2. The framework was greatly influenced, and integrates, elements from two well-known frameworks – Social Ecological Systems Framework (SESF) (Ostrom 2007, 2009) and Ecosystem Services Framework (ESF) (Costanza *et al.*, 1997; Ehrlich *et al.*, 2012; MA, 2005; TEEB Foundations, 2010; Turner and Daily, 2008). The SESF provides rather general, theoretical-conceptual, approach to integrate the interconnections and dynamics between social-economic and ecological systems, as well as among potential key actors and entities driving SES (un)sustainability (governments, banks, firms, households). In contrast, the ESF provides with the empirical basis for the framework and for developing the models, at a lower and more specific level. This was done by integrating and modelling the so-called ES cascade concept (Haines-Young and Potschin, 2010), which links ES providers and ES beneficiaries. The resulting ‘platform’ forms the conceptual framework of this thesis (described in detail in Chapter 2) used to examine what factors and actors are key drivers of SES (un)sustainability in complex SES.

The conceptual and theory building stage was then followed by the core stage of the thesis, based on building one theoretical (Chapter 3) and two empirical (Chapters 4 and 5) models under the conceptual framework in Chapter 2. The objective of the computer models was to answer, through different case-studies, the research questions posed in section 1.2. Based on the research objectives of this thesis and the characteristics of complex SES – comprising multiple scales, feedbacks, stochastic and non-linear processes – Agent-Based Modelling (ABM) was selected as the modelling approach of this research. ABMs have been widely used, not only in relevant areas for this research, such as ecology (Grimm, 1999) and economics (Farmer and Foley, 2009; Tesfatsion and Judd, 2006), but also in many other diverse fields, e.g. sociology (Gilbert and Troitzsch, 2005), geography (Brown & Robinson, 2006) political sciences (Epstein, 2002; Kollman and Page, 2006). The three models (Chapters 3-5) share the same conceptual framework and modelling technique, yet each model was adapted to the particular context of the SES studied. This involved the use of specific information, and data, from the literature, expert knowledge, and the integration of other modelling techniques in addition to ABM, such as Geographic Information Systems (GIS) or Bayesian Belief Networks (BBN). Overall, the main objective of the exploratory quantitative modelling steps was to elicit broad, and simultaneously in-depth, information on SES sustainability, and the factors driving it, for the case-studies selected.

The third stage was based on analysing and integrating the results derived from the three models (Chapter 6). The objective of this stage was two-fold; namely to (i) provide answers for each specific case-study, and (ii) to link and integrate all the results together in order to provide an overall answer to the research objectives of this thesis. Finally, the results analysis and discussion served as a basis to highlight the thesis' contributions and propose future research pathways relevant to this research project.

1.6 Thesis structure

This chapter began by outlining the conceptual underpinnings of the thesis and identifying the challenges within sustainable development. This was then followed by a contextualization of the concepts of sustainability and sustainable development in this thesis, as well as an introduction to the case-study areas and the rationale behind their selection. Next, the research objectives of the research were presented, followed by the research strategy and methodological approach, and finally the thesis structure.

Chapter 2 analyses the literature reviewed, presents the conceptual framework and describes the general methodology of the thesis. Afterwards, the thesis results are presented in three different but related chapters (Chapters 3-5) – each of them presenting its own specific introduction, method, results and discussion sections. The results chapters are followed by a general discussion chapter (Chapter 6) that integrates the thesis results, analyses the research objectives addressed, and explains the thesis contributions to literature, as well as further research. Finally, Chapter 7 presents the epilogue of the thesis, including a brief analysis and future pathways with regard to the concept of sustainable development, followed by a final reflection.

Figure 1.3 shows the PhD thesis outline and thesis structure.

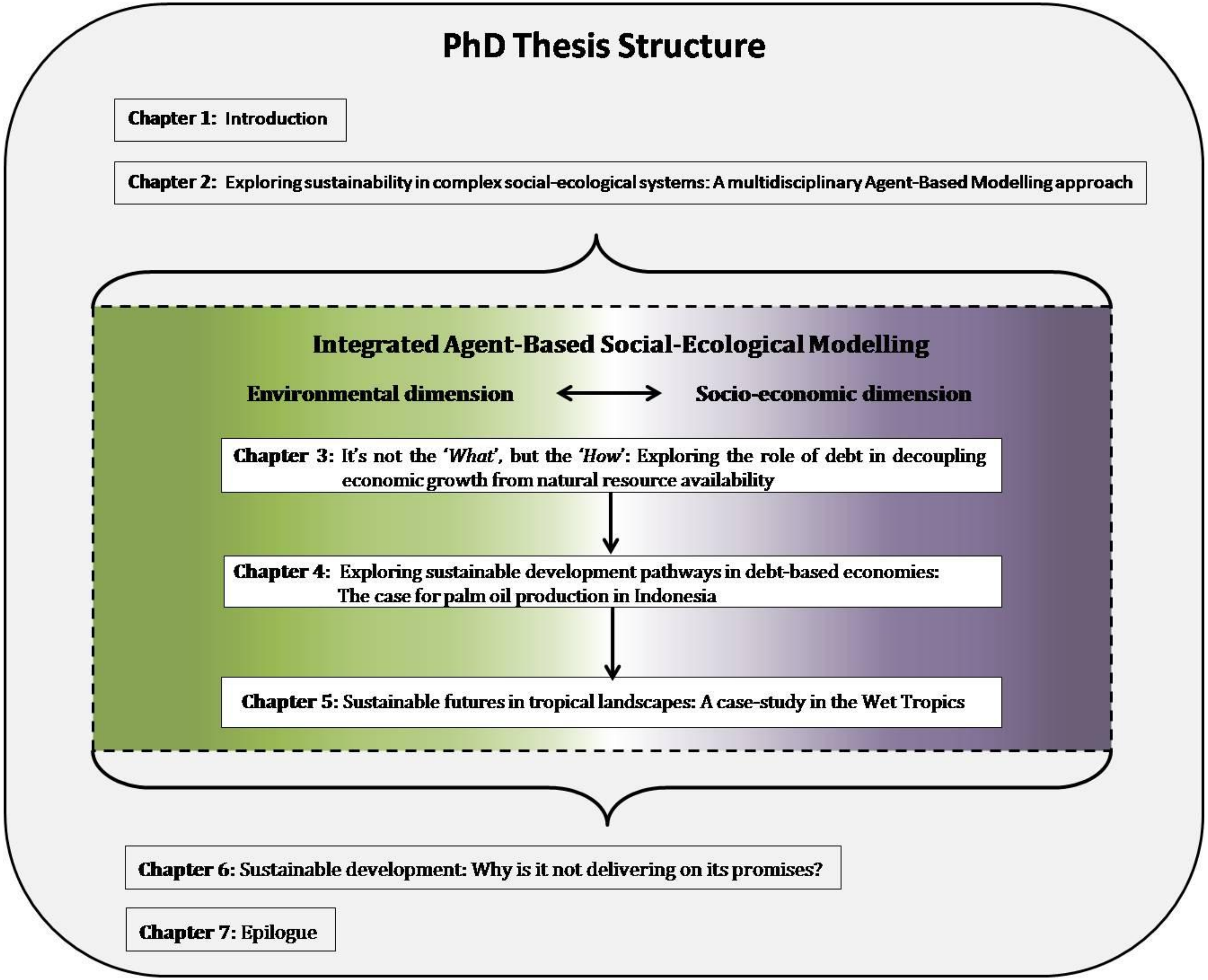

Figure 1.3: Structure of the thesis and relationships between chapters. *Source*: author

More specifically, each chapter includes, and is structured, as follows:

Chapter 2 provides the theoretical and modelling basis of the thesis, in addition to presenting the conceptual framework. First, the historical context and academic literature on sustainable development, economic growth, economic-environmental (de)coupling, externalities, and key actors and entities with regards to (un)sustainable development is presented (section 2.2). This is followed by a description, and background, of the conceptual framework used in this thesis (section 2.3).Finally, the main methodological approach selected – ABM – is analysed (section 2.4). Note that other additional modelling techniques used are specified in the corresponding chapter (i.e. Bayesian Belief Networks and Geographic Information Systems, Chapter 5).

Chapter 3 is the first of the substantive results chapters and applies the conceptual framework presented in Chapter 2. It presents a conceptual ABM focused on exploring whether there is a built-in bias in the current economic system towards unsustainable natural resource use. The model, which includes interactions between banks, firms, households and governments, is built by integrating an environmental system into an ABM representation of Steve Keen's (2009, 2010a) macroeconomic models. The chapter aims to identify which socio-economic and governance factors lead to decoupling scenarios between economic growth and environmental pressures in complex coupled SESs; special attention is given to the relationship between debt and environmental sustainability. The factors driving SES (un)sustainability in this model are explored, as well as broader issues around the role of credit-based economic systems and governments with regards to sustainable development.

Chapter 4 presents an empirical application of the conceptual ABM from Chapter 3, using Indonesia as a case-study. It explores the impacts on SES sustainability of the current debt-growth cycle in Indonesia, the world's largest debt-driven producer of palm oil. In particular, the ABM is built upon empirical data and expert knowledge, and it analyses the impact on SES sustainability of different scenarios (2017-2050) considering different power relations and conflicts among economic forces (land clearing for palm oil production) and conservation forces. The impacts of such interactions are analysed over three main (interrelated) indicators for SES sustainability: food production (palm oil), climate change mitigation (carbon emissions) and biodiversity conservation. This chapter highlights important socio-economic and governance factors with regards to (un)sustainable development in Indonesia. The chapter also provides further understanding of how key global macroeconomic issues (i.e. debt) are entangled with environmental shifts at national scales. Finally, the chapter draws out important sustainability lessons for developing countries that are highly dependent on debt-based production systems.

Chapter 5 explores the impacts on SES sustainability of economic and conservation force dynamics (similar to Chapter 4); yet, this is done under a completely different context. In particular, an empirical and spatially-explicit ABM is presented which uses the Wet Tropics of Queensland, Australia, as a case-study. Here, a land-sharing (LSH) versus land-sparing (LSP) approach is used, suitable for this case-study due to the particular geographic and spatial context of the Wet Tropics region. Furthermore, other modelling techniques are integrated within the ABM, namely BBNs and GIS, together with expert opinion and empirical data. The model examines the impact of economic-conservation force interactions on the same key nexus for SES sustainability: food production (sugarcane), climate change mitigation (carbon sequestration) and biodiversity conservation. The results are used to explore which combination of socio-economic and governance factors drive SES (un)sustainability in the Wet Tropics. The chapter is also used to propose potential pathways that could help limit the expansion of agricultural intensification while improving sustainability in tropical SES.

Chapter 6 discusses the lessons learnt in the results chapters – Chapters 3-5. First, the ABMs built are integrated under a single ontology (section 6.1); it explains and justifies the relationships among model elements and processes, which were all constructed under the same conceptual framework (presented in Chapter 2) and modelling technique. The next section (section 6.2) integrates the results and conclusions from the results chapters and builds upon the results obtained to discuss sustainable development from the viewpoint of the need for a ‘bounded’ economy that integrates natural capital and externalities into the economic system. Afterwards, section 6.3 presents the model and theoretical contributions of the thesis, as well as potential further research. The final section (section 6.4) summarizes the conclusions of the thesis.

Chapter 7 presents the epilogue of the thesis, where section 7.1 outlines the need of a renewal of the concept of sustainable development, and section 7.2 shows some final reflections on the research performed and topic addressed as part of this thesis.

Chapter 2:

Exploring sustainability in complex social-ecological systems: A multidisciplinary Agent-Based Modelling approach

> "*After one look at this planet, any visitor from outer space would say "I want to see the manager"*
>
> – William S. Burroughs (American writer, 1914-1997)

This chapter presents the literature review, conceptual framework and modelling approach of this thesis. Prior to developing the specific conceptual framework for this thesis, a review of the existing frameworks, theories and metaphors with regard to sustainable development was performed. Furthermore, literature was reviewed on the different aspects covered by the thesis, including mainstream economics, ecological economics, social-ecological systems, environmental governance, conservation, ecosystem services, computer modelling. The modelling approach selected is Agent-Based Modelling (ABM), used in each of the results chapters (i.e. Chapters 3, 4 and 5). New insights are provided as modelling outcomes from each of these chapters, which are ultimately synthesized in the discussion chapter (Chapter 6).

This chapter is organized as follows: first, a historic approach to the concept of sustainable development is described, followed by an analysis of the current economic paradigm and the disconnection between the economic system and nature; this is followed by a description of the so-known "market failures" (or externalities) and the role of key system actors (governments, financial institutions, corporations) in this regard. Second, a historic review of the way in which social (economic) and environmental sciences have changed their approach to addressing sustainability issues over the last decades is presented. Finally, the conceptual framework of this thesis is presented, followed by an analysis of the modelling approach used throughout the Thesis.

2.1 Sustainable development: A historic approach

2.1.1 The concepts of sustainability and sustainable development

Since the word “sustainable” first appeared in the 1610s – meaning “bearable” or “defensible” (Online Etymology Dictionary, 2013) – there has been a significant change in its meaning. Many consider Rachel Carson’s book *Silent Spring*, published in 1962, as the turning point in our understanding of the interconnections among the environment, the economy and social well-being (Carson, 1962). However, it was from the 1970s onward when the popularity of the term sustainability increased rapidly, due to rising concerns with population growth, resource consumption and depletion (e.g. wood, coal, oil), and the widespread deterioration of ecological conditions across the globe (Du Pisani, 2006). One of the first official uses of the term sustainable in the contemporary sense was by the Club of Rome in 1972, through the report on the ‘Limits to Growth’, written by a group of scientists led by Dennis and Donella Meadows (Meadows *et al*., 1972). Currently, sustainability is known for its three ‘pillars’, or the ‘triple bottom line’ (Holdren, 2008), which is an approach used to define the complete sustainability problem. This consists of at least the economic, social, and environmental pillars, where the weakness in any one pillar makes the system as a whole unsustainable.

In 1980, the International Union for the Conservation of Nature published a world conservation strategy that included one of the first references to ‘sustainable development’ as a global priority (IUCN, 1980). Two years later, the United Nations World Charter for Nature raised five principles of conservation by which economic development affecting nature is to be guided and judged (UN, 1982). These reports enhanced a shift in the discourse from ‘sustainability’ to ‘sustainable development’, consisting on a more realistic approach that applied the abstract concept of sustainability to the current development paradigm. As a result, the key milestone of sustainable development appeared in 1987, during the ‘Report of the World Commission on Environment and Development – Our Common Future’ (Brundtland Commission, 1987). Here, sustainable development was defined as a:

"Development that meets the needs of the present without compromising the ability of future generations to meet their own needs."

The Brundtland Report moved the concept of sustainable development beyond the initial sustainability framework to focus more on the goal of socially inclusive and environmentally sustainable economic growth (Sachs, 2015). Thus, sustainable development, as defined in 1987, proposed a new path for the society, an innovative and promising idea focused on balancing economic development with the social and environmental pillars. As a result, various reports and conferences took place during the following years highlighting the importance of achieving sustainable development. Thus, in 1992, the United Nations Conference on Environment and Development (UNCED), also known as the Rio de Janeiro Earth Summit, developed the Agenda 21 (UNCED, 1992). Ten years later, in 2002, the World Summit on Sustainable Development (WSSD) (i.e. Johannesburg Summit) took place, followed by the creation of the Millennium Ecosystem Assessment (MA) during the period 2001-2005 (MA, 2005). Finally, in 2015, after the 2012 United Nations Conference on Sustainable Development in Rio (commonly called Rio+20 or Rio Earth Summit 2012), the United Nations General Assembly formally adopted the 2030 Agenda for Sustainable Development. This agenda was based on 17 interrelated Sustainable Development Goals (SDGs), to be implemented and achieved in every country from the year 2016 to 2030 (UN, 2016).

Although there has always been some dissatisfaction with the definition of sustainable development from the Brundtland Report (for example, see Fuentes, 1993; Johnston *et al.*, 2007; Levin, 1993), this concept has become remarkably popular. Currently, more than one hundred variations of the concepts of sustainability and sustainable development exist (Marshall and Toffel, 2005), across different political, industrial, societal and academic domains. The problem is that, due to its popularity, the meaning of the concept has become fuzzier; its malleable nature, which stresses the interconnection of 'everything', has made it vulnerable to distortion by woolly thinking and has become an attractive term for special interest groups (Kates *et al.*, 2005). Similarly, its proliferation has caused it to be frequently employed as a vague gesture to the need for environmental conservation in the context of prioritizing economic growth

(Wu, 2013). Although such vulnerability and blurring of the concept was probably unavoidable, there is a need to turn the SDGs into effective governance and policies throughout the globe. For that purpose, we need to take advantage of the powerful following that the concept has gained over the past two decades. If it recovered its original meaning from 1987, it could become a guiding force for governments, firms, society and non-governmental organizations.

2.1.2 Evolution of the current economic paradigm

In order to address any sustainability problem, including the SDGs, it is necessary to first understand the historic context and nature of the current free-market capitalist, neoliberal economy, as well as the incapability of the current economic system to enhance environmental sustainability and provide public environmental (next section) and social goods (‘market failures’).

The current monetary system, initiated by the Bank of England around 1700 under an exponential growth paradigm (Martenson, 2010), was designed and implemented at a time when the earth’s resources seemed limitless. In 1798, Thomas Malthus postulated that the human population’s geometric growth would exceed the arithmetic returns of the earth at some point in the future (Malthus, 1798); that is, the exponential growth of human numbers would meet with the constraints imposed by a finite world. Currently, it is well-known that an exponential growth rate will not be able to continue before retarding influences set in, such as food supply constraints (Godfray *et al.*, 2010).

After the Second World War, the economies of developed countries started to experience a growing virtuous cycle, with the creation of strong geopolitical unions and development of welfare states through access to cheap energy and other raw materials (HowMuch, 2017). The economic growth was enhanced by the international abandonment of gold settlement in 1971; this process reinforced further economic growth (Herold, 2012) through a banking monetary system focused on continuously providing new loans (debt) that had to be paid back with interest (Martenson, 2010). As a result, from the early 1980s the build-up of this debt-driven neoliberal growth model took off and, thus, enhanced the role of financial actors, markets, and institutions in the operation of the economy, i.e. known as finance-led growth regime or finance-dominated capitalism (Hein and Truger, 2010).‘Debt-led consumption boom’ economies, such as USA and UK, started to dominate the economy; other countries, such as Germany, Japan and China, applied a ‘mercantilist export-led’ strategy, yet these were also dependent on the debt-fuelled growth of the prior countries (Hein *et al.*, 2015). Reduction of barriers to international capital flows and the related trade in complex financial instruments also helped reinforcing the debt-growth cycle. In parallel, other elements of the monetary system besides the total credit market debt started to also exhibit exponential growth rates, e.g. money supply (Federal Reserve Board, 2018b) and household debt (Federal Reserve Board, 2018a). Together with a continuous world population growth – which increased from around three billions in 1960 to more than five in 1990 (World Bank, 2015) – these exponential processes enhanced further money and debt creation. Ultimately, such neoliberal growth paradigm has given rise to a large number of financial crises[6] – culminating in the Great Recession starting in 2007-2008 – as well as the instability of the current economic system (Russo, 2017).

Economic growth can be therefore pictured as a reinforcing loop, similar to a snowball collecting more and more layers as it rolls down a hillside. In the short term, the benefits of economic growth are many: the more that businesses and nations grow and profit, the more individuals have jobs, resources and improved quality of life (Higgings, 2013). However, the need of the economy to maintain an exponential growth does not consider the constraints of the natural laws within which the material and energy systems operate (Hubbert, 1974). Therefore, there is a need to address and integrate the negative environmental impacts exerted by economic growth on development analyses, thus enhancing the sustainability of our environmental life support system.

2.1.3 Economic growth and environmental pressures: A broken marriage?

In 1992, seventeen hundred of the world’s leading scientists argued that the economic system was on a collision course with the natural world (Kendall, 1992). Since then,

[6] Note that not all economic crises can be attributed to the neoliberal growth paradigm, since there were financial crises much earlier than 1980s.

with the exception of stabilizing the stratospheric ozone layer, humanity has failed to make sufficient progress in solving most environmental challenges, such as climate change, freshwater availability, deforestation, marine fisheries collapses, among others (Ripple *et al.,* 2017). The problem lies on the current need of the economic system to consume natural resources in order to grow. For instance, recent studies state that the dependence of global economic growth on natural resources has increased by over 60% during the period 1900-2009 (Bithas and Kalimeris, 2018).This results in an increased commodification of nature and privatisation of commons, as well as the production of waste that pollutes the different ecosystems and atmosphere. Moreover, environmental pressures enhanced by economic activities can have a high monetary cost, such as air pollution across Europe, which costs 1.6 trillion USD a year in deaths and diseases (WHO and OECD, 2015). The situation is compounded by the market deregulation and reduction of international trade barriers, among other aspects of the market economy, which permit financial institutions to expand their activities and acquire more powerful positions in the economy (Hein and Truger, 2010).

Recent research points towards one underlying factor that could be threatening economic development and environmental sustainability: monetary debt (ICSU and ISSC, 2015). Essentially, the never-ending economic growth paradigm requires the accumulation of more and more debt, while future growth – fuelled by ever-increasing amounts of energy and resources – is needed to repay the debt (Daly, 2011). And so the cycle continues. Such increase in debt stocks and debt-driven crises could lead to further illegal logging, unsustainable food production and increasing emissions of GHG, among other sustainability issues (Antoniades *et al*., 2017). One example of this debt-(un)sustainability relationship can be found in Southeast Asia, where more than $45 billion in credits have been loaned out between 2010-2017 by overseas banks to companies operating in different sectors (e.g. palm oil, timber) whose activities are resulting in biodiversity loss and GHG emissions (see Forest & Finance, 2016). The problem is that global debt has now reached historically unprecedented levels (Ciolli, 2018), yet research studying the impact of debt dynamics on environmental sustainability is scarce.

There is a need to advance scholarship on more sustainable pathways to development through *decoupling* economic growth from environmental pressures under debt-based economies. The concept of 'decoupling' is a very recent term, as until the 1970s there was little evidence that economic growth and environmental pressures could be decoupled (Smith *et al*., 2010). According to the OECD (2002), the term decoupling refers to breaking the link between the growth in environmental pressure associated with creating economic goods and services. Thus, decoupling is the objective of separating the economic growth (increase) from environmental impacts/pressures (decrease), so that net 'win-win' scenarios are achieved. Figure 2.1 shows three types of industrial and biological growths, representing different processes with regard to economic growth and/or natural resource use over time.

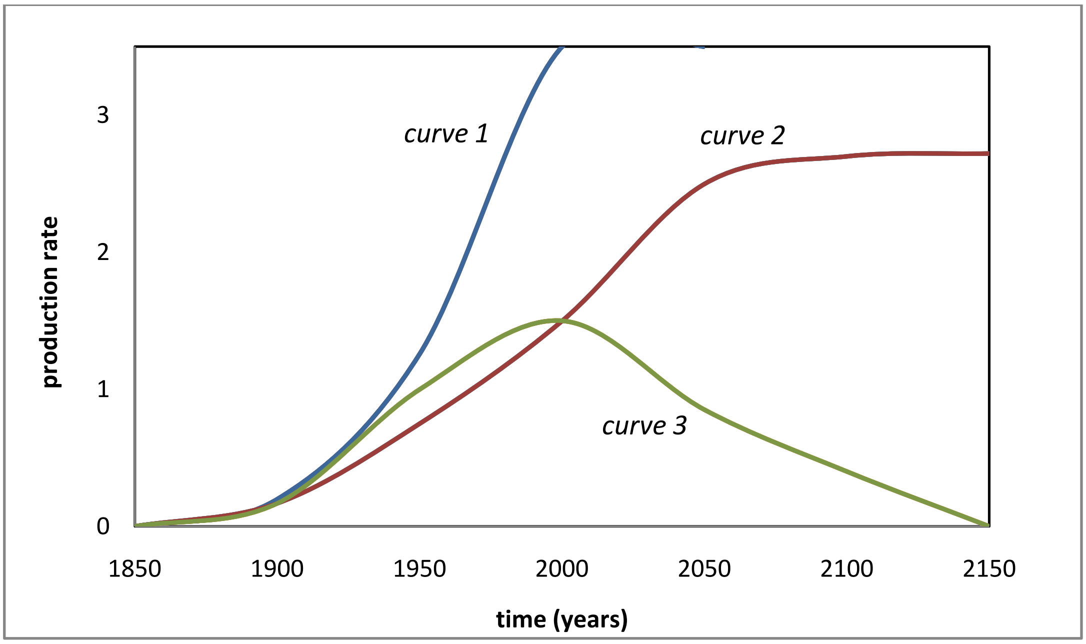

Figure 2.1: Three types of growth (*adapted from Hubbert, 1974*). Curve 1 – exponential, unlimited growth; curve 2 – asymptotic growth in which production meets equilibrium with supply from natural capital; curve 3 – irruptive growth, where there is degradation of natural capital by production.

Hubbert (1974) argued that it is physically and biologically impossible for any economic component to follow the exponential growth phase (*curve 1*) for more than few tens of doublings. On this basis, given more than three doublings since 1850, the exponential phase of the industrial growth and monetary production that has dominated human activities would be now drawing to a close. As a consequence, some industrial components and use of resources should be already leveling off to a steady state (*curve 2*), while some others could ultimately lead to zero (*curve 3*).

Frameworks for achieving economic-environmental decoupling are still in their infancy (UNEP, 2011). Furthermore, there are considerable difficulties in increasing socially desirable goods and services without raising the use of resources or increasing environmental degradation, i.e., climate change, ecological footprint, pollution, waste and reduced biodiversity (Weinstein *et al*., 2013). Therefore, decoupling economic growth from environmental pressures may need more than just materialistic solutions. In this regard, some scholars argue that we need to turn our cultures upside-down in order to nudge human nature away from unsustainable economic growth (Higgings, 2013). This more "subjective" act of disunion and separation refers to the current disconnection between economic and environmental paradigms, i.e. between the economic system and nature, for which a change in the current consumerism and materialistic values would be needed based on opposite and non-compatible objectives between both paradigms. Other scholars argue that we need to combine a cultural change with further technological development in order to achieve decoupling scenarios (Higgings, 2013). Based on the latter approach, a sustainable growth would not be possible without a "dematerialized" development, where technology efficiency permits society to enhance aggregate GDP – or GDP per capita (see Bithas and Kalimeris, 2018) – without depleting natural resources further. Additionally, decoupling economic growth from environmental pressures would probably need an increased energy supply, decreased per capita energy use, decreased consumption, a reduction in human population size (Brown *et al*., 2011), and, overall, an environmental fiscal reform.

Considering the unrealistic idea of reducing environmental pressures to zero, at least in the short-term, the aim should be to reach a lower bound signifying the minimum amount of environmental pressure to deliver the economic growth (Smith *et al*., 2010). For that purpose, contexts where environmentally sustainable investments help generating economic value for both the public and private sectors are necessary (Broadstcok, 2016). There are examples of success stories showing that, with political will, effective and purposeful policies, technical innovation and appropriate management of vested interests, reductions in environmental pressures can be achieved while maintaining strong economic growth. Leading the way are countries such as Sweden – which has committed to be independent of oil imports by 2020[7] – or Costa Rica – which has committed to have a net zero carbon footprint by 2021 (Smith *et al*., 2010), yet the economic context of Costa Rica cannot be compared to other more developed countries. Other past examples include those from the energy sector in industrial countries between 1972 and 1986 (Brundtland Commission, 1987). More specifically, the US economy grew by 27 per cent over a seven-year period starting 1979, while oil consumption and US oil intensity (barrels per dollar of real GDP) fell by 17 per cent and 35 per cent, respectively, during 1977-1985 (Lovins and Datta, 2004). Another example includes the efforts to reduce air and water pollution (Smith *et al*., 2010). In particular, there was an effort to decouple economic growth from SO_2 pollution through the 1983 Helsinki Protocol and the United Nations Economic Commission for Europe (UNECE) Second Sulphur Protocol in 1994. The Second Sulphur Protocol committed nations to targets of reductions of 50 per cent by the year 2000, 70 per cent by 2005 and 80 per cent by 2010. Initial perceptions were that it would be incredibly costly, but the arrival of cost-effective low-sulphur fuel and a range of supporting technologies altered the cost implications such that the use of sulphur could be reduced for significantly less cost than originally anticipated – US$90 per tonne rather than the anticipated US$1000-1500 per tonne. In this case, economic growth and the reduction of environmental pressures, i.e. the emissions of sulphur dioxide, were compatible, along with reductions in nitrogen oxides (NO_x) and fossil fuel consumption.

Despite the lack of mechanisms of the current economic system to self-regulate long-term sustainable planning of public goods, the world has (still) sufficient stocks of natural capital to meet most of society's demand at the current time. It is therefore important that these and other decoupling examples around the world, as well as the role of key actors and institutions driving these processes, are understood and analysed, thus

[7] Note that some authors argue that achieving (or not) sustainability is a matter of scales, and that the issue of sustainability displacement should be considered. Based on this idea, the achievement of sustainability can be shifted to some other place and future time, rather than being delivered in the here and now (see Saunders and Hughes, 2018). This is further discussed in Chapter 6 of this thesis.

proposing strategies at the regional and national levels that can achieve strong decoupling targets.

2.1.4 Externalities and market failures

The capitalist system fails to adequately address the environmental impact costs and to value natural capital, i.e. the stock of natural resources that combine to yield a flow of benefits (i.e. Ecosystem Services, ES) to people (WBCSD, 2017). As human populations grow, and grow increasingly disconnected from nature, achieving sustainable development will not be possible without understanding how the economic system affects natural capital – and, therefore, our long-term wellbeing – and how to integrate natural capital into the economic system, including policy, decision-making (WBCSD, 2018). In most cases, an old – yet key – dilemma preventing positive decoupling scenarios is based on the incapacity of the market economy to efficiently integrate and account for externalities. Externalities constitute a form of “market failure” in the form of costs or benefits of an economic activity that are experienced by unrelated third parties (Gies, 2017). The classic example of a negative externality is a factory that dumps effluent into a river. Unlike homeowners who pay for garbage pickup, the factory’s owners pay nothing for disposing their waste into the river. But humans and other creatures living downstream do pay a cost, while cities have to build expensive treatment plants.

Externalities are more common when public goods, or commons, are involved, which are defined as being non-exclusionary (i.e. individuals cannot be effectively excluded from use) and non-rivalrous (i.e. consumption by individuals does not reduce quantity or availability to others), e.g. clean air, clean water, biodiversity, fish stocks (Cornes and Sandler, 1986). Commons are free goods, produced by nature and available to everybody. These are estimated to be worth more than the entire world's private assets combined (see Costanza *et al.*, 1997), with public goods usually subject to ill-defined property rights, resulting in society not placing enough value on them. The market economy does not integrate and account for externalities for one very simple reason: intervention to protect those realities is counter to economic development or, for that matter, means incurring high (monetary) costs (Helbling, 2010). In this regard, neoclassical economists – who recognize externalities as a form of “market failure” – support government interventions to correct for the effects of externalities when the market failure is detrimental to society or environment. The power of governments could be therefore used to force the market to account for costs that would otherwise not be included (DeNyse, 2010), for instance by establishing institutional frameworks that allow for proper bargaining among parties involved in externalities (Helbling, 2010).

A well-known mechanism to internalize externalities is based on market-based, self-correcting regulations, which are cost effective mechanisms that encourage technological progress (Labandeira-Villot, 1996). Examples include taxes and subsidies, such as ‘green’ financial instruments, i.e. ‘green bonds’, which offer the opportunity to finance projects that generate financial profits and environmental benefits (UNDP, 2018). Another mechanism that has gained popularity over the past years is the tradable emissions permits (DeNyse, 2000). International and regional carbon markets, such as the European carbon market (EU ETS), were created to help to reduce the rate of climate change in the long term (Chichilnisky and Sheeran, 2009). Other mechanisms to internalize externalities include auction development rights – where the government places itself as a market participant and avoids over-exploitation and under-valuation of natural resources, e.g. countries in Africa and South America charge fishing trawlers a fee for the right to fish in their waters (DeNyse, 2000); or the integration of natural capital in the Gross Domestic Product (GDP) – since capitalism neglects to assign any value to the natural capital on which it depends. Finally, ecosystem service-based approaches have also been considered as frameworks that could help integrating ecosystem services (ES) – the benefits that humans obtain from nature – and the ecosystem structure that generates them into the market system. One approach argues that ES should be treated as market commodities, either by estimating their monetary value and including that signal in market prices or decisions, or else by making the resources excludable commodities subject to market allocation (Gies, 2017). In particular, considerable attention has been given to the monetary valuation of non-excludable resources over the past decades (e.g. Costanza *et al*., 1997; Getzner *et al*., 2005; Pearce and Turner, 1990).

Internalizing externalities requires synergies between governments, business and the financial sector, considering that the latter two are responsible for most part of the degradation of natural capital worldwide. However, conflicts of interests among these actors often results in unsustainable economic growth is imposed over environmental conservation. Therefore, research in the interplay between these actors and environmental sustainability is necessary if the aim is to create future sustainable scenarios showing decoupling contexts.

2.1.5 Governments, markets and financial institutions: analysing key actors for (un)sustainability

Despite the importance of internalizing externalities for global sustainability, there are well known problems and obstacles at the time of implementing the above-noted mechanisms. For example, defining property rights, uncertainty (who is responsible for damages?), high transaction costs (Helbling, 2010), measurability and monetary valuation of unmeasurable goods (e.g. cultural ES; biodiversity) (Small *et al*., 2017), among others. Yet, one of the most important obstacles to enhance environmental sustainability and decoupling processes is related to conflicts of interest between governments and private-financial institutions. Banks, investors, and other financial actors play an important role in the global economy, which itself is a prime driver of ecological change (Galaz *et al*., 2015). More specifically, financial markets and actors drive land and ecosystem change under complex and multilevel contexts (Berkes *et al*., 2006; Lambin and Meyfroidt, 2011), thus affecting ecological systems significantly. Examples when large investors or banks have failed to consider and address large-scale ecological risks are numerous. For example, in 2014, the Deutsche Bank organized an initial public offering for China Tuna Industry Group Holdings, one of China's largest tuna longline companies. The expansion plan of the Chinese company, however, was revealed to be based on incorrect fish stock data that far exceeded existing Bigeye tuna stocks in the region (Winner and Associates, 2014). As a result, China Tuna had to withdraw the offering from the Hong Kong Stock Exchange, which did not only come with environmental impacts, but also reputational risk and negative financial consequences (UNPRI, 2011).

Enhancing environmental sustainability under the current economic paradigm requires governments to counterbalance the profit-seeking behaviour of financial institutions and business – which solely focus on gaining profits and economic growth – through different strategies and policies (Abel *et al*., 2006). The previously mentioned case in Southeast Asia – where overseas banks fund unsustainable agricultural and forest production through debt (Forest and Finance, 2016) – is one of the many examples where government intervention could, through strong public governance and legislation, counterbalance such unsustainable practices. However, the current weak public governance in some developing countries is not enough to reduce the power of financial institutions and, therefore, halt the negative effects driven by the latter on the environment, e.g. Indonesia (OECD, 2016). Hence, the sustainability problem arises sometimes from the political difficulty of implementing government policies that would, indirectly, reduce the power of influential financial institutions (Abel *et al*., 2006), such as commercial banks. Most economic actors are not interested in any paradigm shift that may reduce their profits, and this is why governments are not usually free to invest or create policies that play against the interests of industries and other interest groups (Abel *et al*., 2006). This could be one of the reasons for the difficulty of decoupling economic growth from environmental pressures, as well as the reason why systems so often remain maladapted to current unsustainable conditions, to the point of collapse.

While there are studies exploring similarities between complex economic and ecological systems (May *et al*., 2008), few scholars have studied the intricate interplay between the two systems. Examples include analyses on how international trade drives ecological change in land- and seascapes (e.g. Berkes *et al*., 2006; Lambin and Meyfroidt, 2011), the value of biodiversity and 'natural capital' (e.g., Costanza *et al*., 1997; Turner and Daily, 2008), or the potential for new financial instruments to increase private and public investments in conservation and ES restoration (e.g., Chichilnisky and Heal, 1998; Loucks and Gorman, 2004). Thus, there is a need to further understand the links between economies and financial markets with ecosystems, particularly considering the role of power-conflicts and power (im)balances between governments, corporations and financial institutions (e.g. investment banks) on enhancing social-ecological (un)sustainability.

2.2 Structured research to study sustainability in complex social-ecological systems: A conceptual framework

In order to address the research aim and objectives posed in section 1.2, a systemic, holistic and interdisciplinary understanding of the interrelation between the economy and the environment in each of the SES modelled is needed. This section presents the conceptual framework of this thesis, preceded by a review of the literature on conceptual frameworks and approaches addressing sustainability issues developed over the last decades until today.

2.2.1 Framework background: The way towards more integrative and interdisciplinary approaches to address sustainable development.

The social science literature shows early examples of human-ecosystem frameworks based on integrating ideas and approaches from ecological sciences into social sciences, such as sociology (Duncan, 1961, 1964; Field and Burch, 1988) and anthropology (Vayda, 1969; Watson and Watson, 1969). However, much of development in natural resource management science since around the 1970s was based on classic utilitarian approaches, which was limited in the sense environmental and social problems were treated in isolation (Berkes and Folke, 1998). Critiques aroused with regards to the simplistic foundations of policy and science on natural resource management, calling for more complex, intellectual tools that could alleviate the excesses of classical approaches to manage resources (Ostrom, 1990). As a result, literature started to show examples of systems-oriented, wide-scope approaches, which considered linkages and feedback processes between systems (Holling, 1978; Walters, 1986). Such emphasis on interdisciplinary, ecological economics approaches to sustainability also emphasized the need for changes in institutions and property rights, e.g. Ostrom (1990) on institutions and collective action; Hanna, Folke and Maler (1996) on property rights; Berkes (1989) on community-based resource management.

Existing social and political science methods and ideas were being incorporated into ecological approaches. For instance, the term “human ecosystems” (Machlis *et al.*, 1997) or “social ecological system” (Redman *et al.*, 2004) started to be included in the literature, so as to emphasize the interaction of the forces acting in these two domains.

However, what it was ultimately needed was a new integrative ecology that explicitly incorporated human decisions, cultural institutions, and economic systems (Grimm *et al*., 2000; Michener *et al*., 2001). With first principles dated back to the first two thirds of the 20th Century (Soddy, 1926; Boulding, 1966), given impetus by more recent work (e.g. Costanza, 1991; Jansson *et al*., 1994), the interdisciplinary discipline of ecological economics was eventually developed as a scientific discipline. Ecological economics is a transdisciplinary discipline focused on developing an economics that is fundamentally ecological in its basic view of the problems; it recognizes the interrelatedness and interdependence between human society and the environment (Costanza, 1989; Costanza *et al*. 1997; Daly and Farley, 2004; Turner *et al*., 1993). Further interdisciplinary disciplines were also developed; for instance sustainability itself created its own field – sustainability science – focused on the dynamic relationship between society and nature at local, regional, and global scales (Bettencourt and Kaur, 2011; Clark and Dickson, 2003; Kates, 2011; Kates *et al*., 2001; NRC, 1999).

Due to this transformation of the science studying sustainability issues, socio-economic and ecological systems were considered linked systems of people and nature, emphasizing that humans should be seen as a part of, not apart from, nature (Berkes and Folke, 1998). As a result, what previously had been divided into “natural” and human systems, it was finally considered a single, complex social-ecological system (SES) (Redman *et al*., 2004). In the current literature, SES are considered to be coupled human-natural systems, characterized for being complex, dynamic, adaptive, interactive and multi-scalar systems (Machlis *et al*., 1997; Redman *et al*., 2004). In this regard, different conceptual frameworks and metaphors have been developed to structure research on sustainability of SES (Redman, 1999; Holling and Allen, 2002; Newell *et al*., 2005; Ostrom, 2007, 2009; Pahl-Wostl, 2009; Scholz, 2011). These outline and predict the links between social, ecological, and economic systems, and thus the dynamics and complexities that hide behind real world sustainable development challenges. Examples include multidisciplinary research (Janssen and Goldsworthy, 1996), Resilience Theory (Gunderson *et al*. 2002ab), Planetary Boundaries (Rockström *et al*., 2009), Ecosystem Services Framework (ESF) (Costanza *et al*., 1997; MEA, 2005; TEEB Foundations, 2010), Ostrom’s Social Ecological Systems Framework (SESF) (Ostrom, 2007, 2009), the IPBES Conceptual Framework (Diaz et al., 2015).

Frameworks differ significantly in their goals, their applicability and their temporal, social, and spatial scales. Therefore, it is difficult to find the perfect framework that works in all settings (Ostrom, 2007). Just as there is no perfect framework, there is no ideal entry point for carrying out analyses of SES (Ostrom, 2007); rather, the entry point depends on the research questions being addressed (see Chapter 1, section 1.2). Thus, selecting one single disciplinary background and conceptual framework may not do justice to the complexity of real-world systems (Pahl-Wostl, 2009). The following sub-section presents the framework of this thesis.

2.2.2 Exploring sustainability in social-ecological systems: A conceptual framework

This section presents the conceptual framework built for this thesis (Figure 2.2), which is used to develop each of the models to be presented in the following chapters. More specifically, the thesis aims to (i) study how different conflicting economic and conservation forces affect sustainability through LUC in different SES, and (ii) analyse which socio-economic and governance factors could create future sustainable scenarios in those SES explored. The first aspect is addressed in the results from Chapters 3-5; in particular, throughout three different models – one conceptual (Chapter 3) and two empirical (Chapters 4 and 5). Finally, the second aim will be addressed in the discussion chapter (Chapter 6), where the Results obtained from Chapters 3-5 are integrated in order to answer the research questions posed in section 1.2 (Chapter 1), as well as to contribute to the literature in SES, sustainability science and ecological economics.

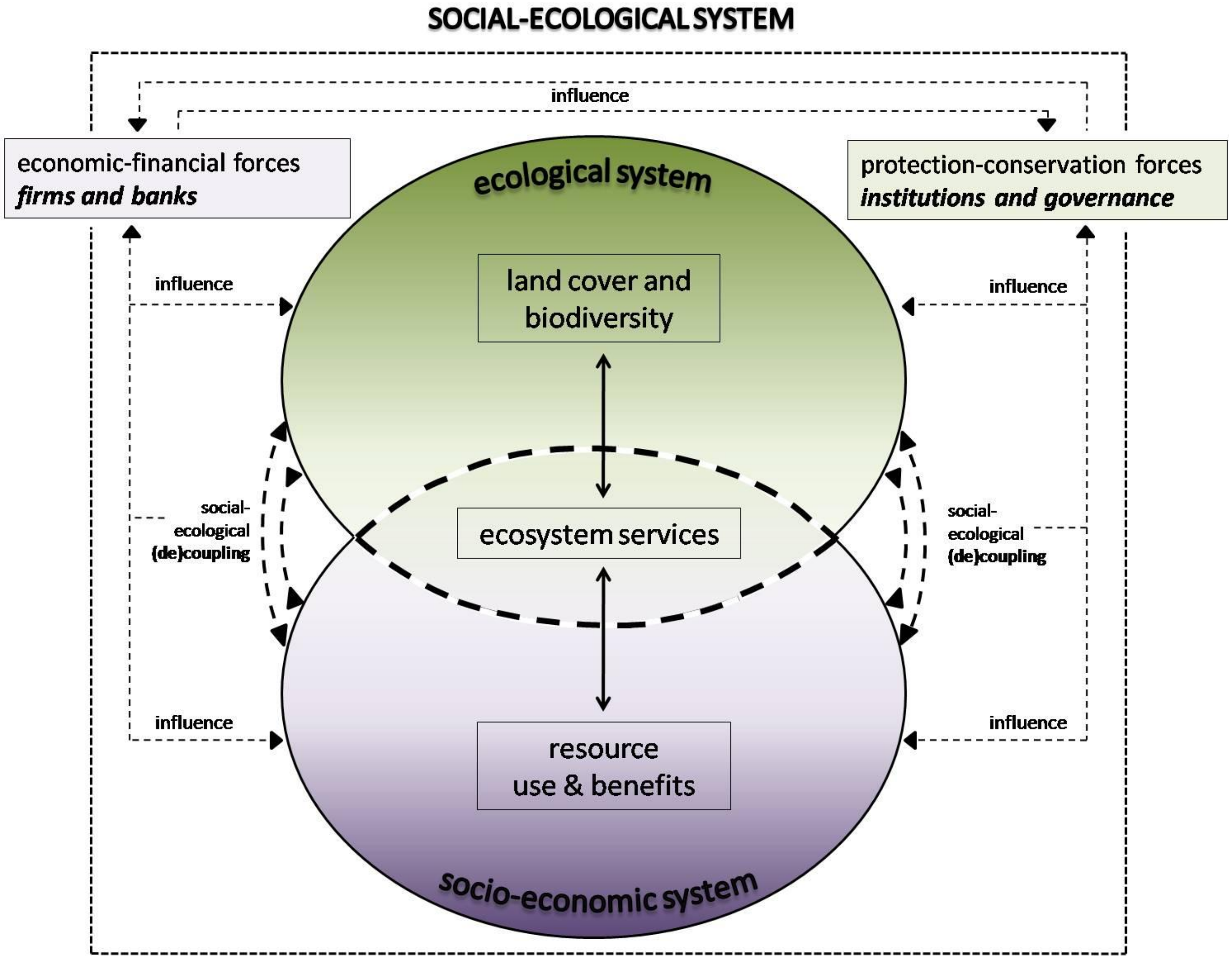

Figure 2.2: A conceptual framework to study the grade of sustainability in SESs. The SES consists of an integrated ecological and socio-economic system. Economic and conservation forces (positioned inside and outside the SES boundary, thus representing both inner and outer forces) drive land-cover change and affect biodiversity; this originates ecosystem services trade-offs and synergies that have an implication for ecosystem service beneficiaries. The processes occurring within the SES affect also economic and conservation forces back – represented by the bi-directional arrows –, while both economic and conservation forces are also linked and influenced by each other. The dashed-shaped pointed oval in the centre represents the decoupling between socio-economic and ecological systems. The grade of (de)coupling between both systems is represented by the dashed arrows to the sides of the dashed oval, which state the extent to which economic and conservation forces de-couple (outer arrowheads) or re-couple (inner arrowheads) the ecological and socio-economic systems. *Source*: author.

The framework is explicitly applied to each of the result chapters, i.e. Chapters 3, 4 and 5. As shown by Figure 2.2, each model includes an ecological system and a socio-economic system, where the ES flow in both directions, including feedbacks and nonlinearities. Economic and conservation forces – driven by financial and governance powers, respectively – drive land cover change and affect biodiversity, which consequently affect the capacity of the land to provide different ES. Finally, this affects the socio-economic context of the system (i.e. ES beneficiaries) and the financial capital of different users. This process is bi-directional, which means that ES users' decisions affect biodiversity and the capacity of the ecological system to deliver different ES. Similarly, both economic and conservation forces influence each other directly – as represented by the arrows on top of Figure 2.2 – and are also affected by the state of the SES itself – see the bi-directional arrows coming into both forces. Note that conservation and economic forces are located both inside and outside the SES boundary, thus representing inner and outer (to the system) forces affecting SES sustainability. Eventually, ES and biodiversity are used as linkers between the socio-economic and ecological systems, as well as indicators to study the sustainability of the SESs analysed. The dashed-shaped pointed oval in the centre of the SES represents the disconnection between socio-economic and ecological systems, and the grade of (de)coupling between both systems is represented by the dashed arrows to the sides of the dashed oval. These arrows state the extent to which economic and conservation forces de-couple (outer arrowheads) or re-couple (inner arrowheads) the ecological and socio-economic systems.

This framework does not present a paradigm-shift with regard to other previous frameworks built to study SES over the past two decades (see Binder *et al.*, 2013)[8]. Rather, this framework is specifically tailored for this thesis by considering the particular nature of the research questions posed and the methodological approach

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[8] The Driver, Pressure, State, Impact, Response (DPSIR) framework, The Ecosystem Services (ES) framework, The Earth Systems Analysis (ESA), The Human-Environment System (HES) framework, The Material and Energy Flow Analysis (MEFA/MFA) framework, The Management and Transition Framework (MTF), The SES framework (SESF), The Sustainable Livelihood Approach (SLA), The Natural Step (TNS) framework; and The Vulnerability framework (TVUL).

selected. In this regard, this framework was inspired by, and includes characteristics from, two well-known SES frameworks; namely the Social Ecological Systems Framework (SESF) and the Ecosystem Services Framework (ESF). Regarding the SESF, it grew out of the recognition that social-ecological outcomes are the product of complex interactions among diverse actors, institutions, and biophysical systems (Agrawal, 2003). Under this framework, a SES is defined as a unit possessing at least one environmental commons (e.g. resources, ecosystem, pollutants), a governance system, and an actor group. The SESF is, therefore, an extensive multitier of a hierarchy of variables that have proven to be relevant for explaining sustainable outcomes in the management of forestry, fishery, and water resources (Ostrom 2007, 2009), and has been used to frame some of the most scientifically relevant issues in SES analyses, e.g. Hardin's (1968) "Tragedy of the Commons". The characteristic tier categorization of the SESF – i.e. resource system (*RS*), resource units (*RU*), governance system (*GS*) and actors (*A)* – was integrated in the framework of this thesis by including environmental commons (i.e. ES and biodiversity), a governance system and an actor group (i.e. ES beneficiaries). Furthermore, the SESF was selected as a basis to build the conceptual framework due to it being considered the only framework that treats the social and ecological systems in almost equal depth (Binder *et al.*, 2013). This is represented, in Figure 2.2, by a SES system composed of an ecological system and a socio-economic system, where none of them takes, in principle, control over the other, and are treated and modelled in equal depth.

On the other hand, although the SESF provided a theoretical basis to the conceptual framework, it was not able to show a straightforward platform or mechanism to directly model SESs through ABM. In particular, a specific link among *land cover and biodiversity–ES–ES beneficiaries* was missing, necessary to build the models of this thesis – where land covers could be represented by patches, ES beneficiaries by agents, and ES as linkers between the latter. Due to this, further characteristics from the ESF were integrated in the conceptual framework. The ESF states that Earth's lands and waters, and associated biodiversity, can be seen as a natural capital stock from which people derive vital ES; these include the production of provisioning services (e.g. food, timber), regulating services (e.g. water purification, crop pollination), cultural services (e.g. inspiration, recreation), and supporting services (e.g. genetic diversity) (Costanza *et al*., 1997; MEA, 2005; TEEB Foundations, 2010; Turner and Daily, 2008). Related to the conceptual framework of this thesis, a key characteristic of the ESF is based on the *ecosystem–ES–users* relationship (or cascade), where ecological systems produce different ES that are ultimately used by different actors (firms, households). The ESF, therefore, provides the basis to model the above-noted relationship, where ES are considered the 'linkers' between the socio-economic and ecological systems (see the *land cover and biodiversity–ES–ES beneficiaries* link in the center of Figure 2.2). Thus, these three elements served as a basis to implement the conceptual framework in each of the models through agents and patches (see Chapters 3, 4 and 5). Furthermore, the ESF facilitated the process of placing an economic value to the benefits (i.e. ES) that different actors (firms, households) obtain from nature. This allowed tracking the impacts on monetary capital of those economic and conservation forces driving LUC and, therefore, affecting biodiversity and the provisioning of different ES.

In conclusion, the framework of this thesis (Figure 2.2), inspired from both the SESF and ESF, was built in line with the nature of the research aim and objectives proposed, as well as the modelling approach used (see next section). In particular, addressing the research objectives required a framework that was able to embrace different dimensions through interdisciplinary research, as well as a suitable context to explore both emergent (bottom-up) and top-down dynamics typical from complex systems. I argue that the framework presented in this thesis is able to assess those variables, at multiple scales across the biophysical and social-economic domains, affecting sustainability of SES over time.

2.3 Agent-Based (Social-Ecological) Modelling

2.3.1 Why modelling?

Over the last three decades, computer models have been used to analyse everything from inventory management in corporations to the performance of national economies and the interplay of global population, resources, food, and pollution (Morgan, 2017). Certain computer models, such as *The Limits to Growth* (Meadows *et al*., 1972), have been front page news. As computers have become faster, cheaper, and more widely available, computer models have become commonplace in forecasting and public policy analysis, especially in economics, energy and resources, and other crucial areas (Sterman, 1991).

What is really the point of computer modelling? It should be remembered that we all use models, such as mental models, to make decisions and solve problems in a daily basis. Anyone who ventures a projection, or imagines how a social or environmental dynamic – e.g. migrations – would occur, is running a model (Epstein, 2008). Mental models are representations of our present understanding of the overall system of interest and are an important first step in problem formulation (Walker *et al*., 2006). Our society is built upon mental models; for instance, belief structures are transformed into society and economic structure through institutions, which represent both formal rules and informal norms of behaviour (Ostrom and Janssen, 2005). However, mental models are typically an implicit model in which the assumptions are hidden, the consistency is untested, the logical consequences are unknown, and the relation to data is unknown (Epstein, 2008). Thus, while mental models are the internal representation of individuals' interpretation of the environment (Ostrom and Janssen, 2005), computer models are external (to the mind) mechanisms individuals create to structure, order, test and explore the environment. The value of computer models derives from the differences between them and mental models, where computer models can improve the mental models upon which decisions are based and contribute to the solution of the pressing problems we face. Thus, the relationship between mental models and computer models is an intimate one, where the latter able to represent the prior in a more efficient, faster and complex way.

The principal result of the increasing use of computer models seems to be, not an improvement in the quality of decision making, but rather a growing sensitivity to the short-comings of models (Bankes, 1993). One short-coming regarding traditional modelling techniques and approaches is related to deductive modelling. Deductive modelling comes from following the logical or mathematical implications of a series of processes to produce predictions about behaviour (Chattoe, 1996). Science throughout the 20th century was dominated by use of a deductive model of explanation, which implied simplifying assumptions such as the modelling of entities as homogenous aggregates (assuming that all actors within a system or group are identical) (Millington *et al*., 2012). Such simplifications were useful before advances in computing model came in, thus being well suited to scientific fields where hypothesis could be constructed (Millington *et al*., 2012). Another short-coming with regards to traditional modelling techniques is related to the use of instrumental mathematical approaches in economic modelling and policy-making. During the 20th century, the most common deductive modelling technique was the solution of sets of differential equations, which basically replaced the economist using pencil and paper with a computer programme. Economists have an unusually strong commitment to utility functions that suffice as a (mathematically) meaningful interpretation of the system being studied (Della Porta and Keating, 2008). Thus, modelling is understood as a mechanical deductive approach to utilitarianism and individualistic rationale choice (Della Porta and Keating, 2008).The reason for the deductive (mathematical) dominance in economics is difficult to explain, yet Chattoe (1996) provided few explanations in this regard. First, economics has been obliged to create a niche for itself as a respectable academic discipline, among other historically more reputed disciplines such as chemistry, physics or philosophy. One way to increase formality and gain reputation was to associate economics with high status physics rather than with other low status disciplines, such as social sciences. In particular, in the early stages of economics, there was considerable enthusiasm for the elegance of Newtonian mechanics as a scientific metaphor. This resulted in the development of theories in which social and economic actors, like atoms with no internal structure, collide in trade driven by the simple acting laws of supply and demand. This could be one explanation why a Newtonian view of the market economy seems to underpin mainstream economics, regardless of other more complex scientific theories and areas existing, e.g. Quantum Mechanics, Thermodynamics, Relativity. Related to this, one other argument for the dominance in economics of simplistic mathematical models argues that mathematical precision was favoured in order for those economists with knowledge of mathematics to gain reputation and advance themselves in science. This was related with the high reputation of mathematics, for instance within physics.

Despite the practical advantages of instrumental uses of computers in modelling, mathematical representation of the dynamics of social and other complex systems is, at least, limiting (Della Porta and Keating, 2008). For instance, economic models based on differential equations are suitable to provide mathematical solutions. However, complex dynamics are not tractable under these approaches, thus modelling complex systems requires techniques that can simulate the different cross-scale, non-linear processes characteristic of such systems (Axelrod and Cohen, 2001; Holling *et al*., 1998). In this regard, the interest in simulation modelling has been increasing in the social, environmental and economic sciences (Barth *et al*., 2012). Simulation should be seen as a technique that is capable of representing a broader class of processes and relationships than the mathematics commonly used in economic modelling. As computing power has rapidly increased, simulation modelling frameworks that improve the understanding of how macroscopic patterns and outcomes emerge from interactions between heterogeneous entities at more disaggregated levels of organization have increased (Epstein, 1999; Grimm *et al*., 2005; Brown *et al*., 2006). The increasing use of computer simulation enhances the possibilities for understanding spatio-temporal dynamics of social and environmental systems (Millington *et al*., 2012). Literature in simulation, on the other hand, shows various methodological debates, including the issue of establishing standards for simulation modelling (e.g., Grimm *et al*. 2006), the discussion whether simulation mainly aims at prediction or at explanation (Epstein 2008), and the challenges of presenting simulation models and their results (Axelrod 1997). Nevertheless, given the method's relatively young age, ongoing methodological debates are to be expected. It can even be considered as a necessary step towards establishing clear methodological standards.

2.3.2 Why (Agent-Based) Modelling?

Different modelling techniques permit the representation of complex SES from different perspectives. Figure 2.3 outlines an adapted decision tree from Heckbert *et al*. (2010) that determines the type of complex systems modelling approach to use for a given application. In attempting to describe SES and other complex systems, equation-based models, systems dynamics, and statistical techniques have been used to good effect. Bayesian Belief Networks (BBN), evolutionary models, and system dynamics are also capable of representing decision making, behaviour, adaptation, and other complex dynamics. ABM, on the other hand, involves autonomous decision makers interacting.

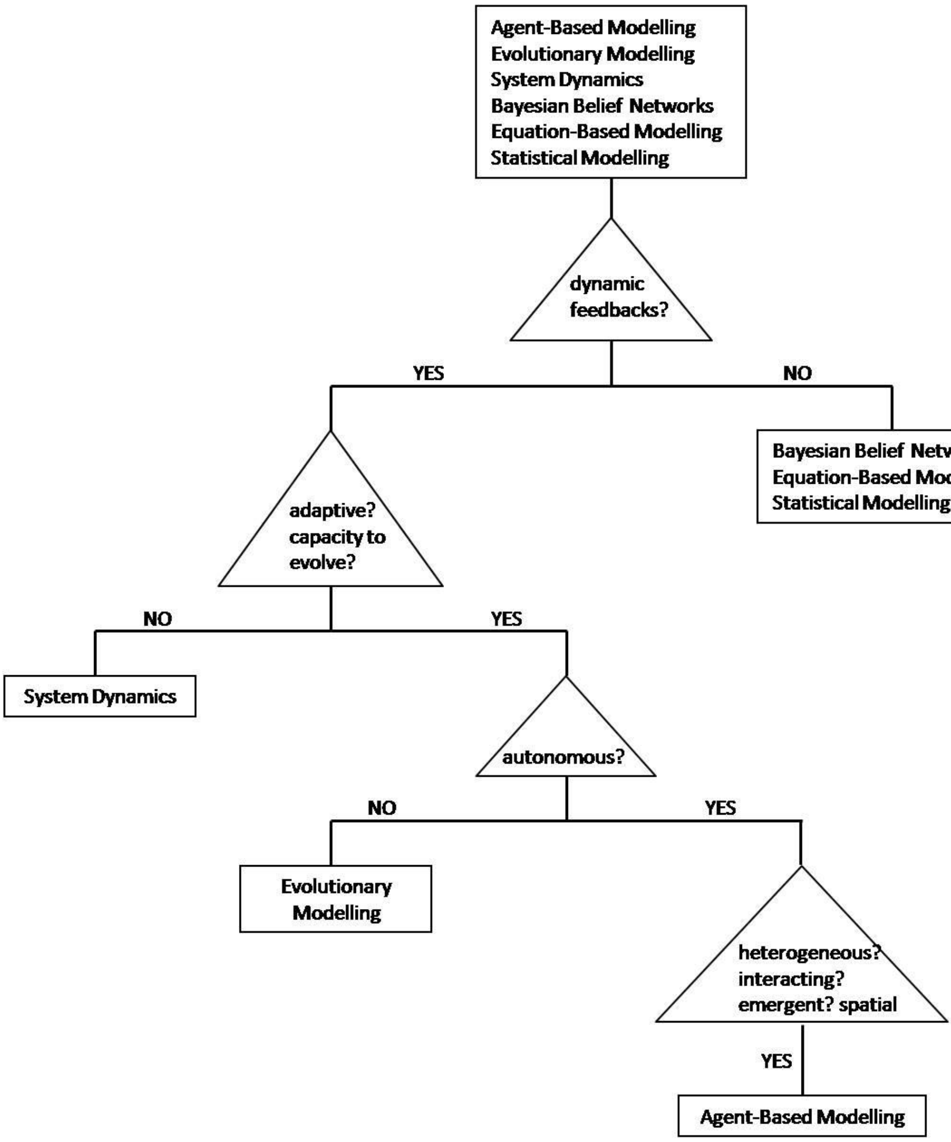

Figure 2.3: Decision tree for selecting modelling techniques to model complex systems, adapted from Heckbert *et al*. (2010). *Source*: author.

Statistical approaches, as well as equations and Bayesian techniques, are powerful ways to characterize complex systems' aggregate attributes and relationships. Since micro-dynamics are implicitly represented in these modelling approaches, they are not capable of providing dynamic feedbacks. Thus, they are at a disadvantage when the subject of the model is not a homogenous population. This is the case, for example, of firms with different financial contexts, resource extraction rates, etc., such as those modelled in Chapters 3 and 4.Top-down modelling approaches, such as BBN, can be useful techniques to complement bottom-up simulation approaches, thus helping to address uncertainties, and incorporate qualitative information and behaviour alongside quantitative data and statistical distributions (Minana, 2016). For example, integrating BBN models into bottom-up modelling techniques, such as ABM, could be an option to compensate the inability of BBNs to easily represent feedbacks and micro-dynamics (described and implemented in Chapter 5). Thus, the flexibility and capacity of most ABM platforms to incorporate not only BBNs, but also equations and other statistical techniques, can be useful to more accurately represent complex systems, whereas the converse is not always the case.

There are top-down modelling approaches that are able to represent feedbacks and describe macro-level processes and complexity. System dynamics models, for instance, possess these characteristics, without having to seek the equilibrium results expected in equation-based models. System dynamics is certainly the most used modelling tool for complex systems, and ecological economics has benefitted in the ability to develop modular system dynamics components connecting phenomena that typically are treated in isolation in some disciplines. However, system dynamics models often include aggregate variables and parameters, thus missing the decisions and actions of multiple individual actors, as well as potentially multiple spatial relationships. Moreover, pure system dynamics models are fundamentally not adaptive and their ability to evolve is limited to variations in parameter values. Thus, the capacity of system dynamics models to micro-dynamics and disaggregate features is limited, yet they can be used more directly to explain macro-level characteristics. These characteristics make System dynamics modelling an inappropriate modelling technique to explore the sort of complex sustainability issues addressed in this thesis.

On the other hand, ABM explores how interactions between agents generate the property of emergence, by “growing” patterns that characterize systems (Epstein, 2006). ABM enables the explicit representation and explanation of adaptive decision making, thus providing an opportunity to explore sustainability issues characterized by heterogeneity, feedbacks through interactions and adaptation (Heckbert *et al*., 2010). The benefits of ABM over other modelling techniques can be captured in four statements (Bonabeau, 2002): (i) ABM captures *emergent* phenomena; (ii) agents are *heterogeneous*, which allows simulating complex and nonlinear behaviour as well as limiting agent rationality; (iii) ABM provides a dynamical natural description of a system or the process under study, rather than only the final output results; and (iv) ABM allows to include social networks and physical space-based interactions, which is difficult to account for with other modelling approaches. As a result ABMs produce a rich set of multidimensional data on macro-phenomena, comprising a wide range of details on micro-level agent choices and their dynamic interactions at various temporal and spatial resolutions (Lee *et al*., 2015). Due to this, ABM has been receiving significant attention recently, being widely employed across fields that are as diverse as biology (Politopoulos, 2007), business (North and Macal, 2007), economics (Tesfatsion, 2005; Farmer and Foley, 2009),education (Abrahamson *et al*., 2007), geography (Brown & Robinson, 2006), health care (Effken *et al*., 2012), medical research (An and Wilensky, 2009) political sciences (Epstein, 2002) and sociology (Gilbert and Troitzsch, 2005). Furthermore, ABM is currently also being used in organizational studies (e.g. Chang and Harrington, 2006), governance (e.g. Ghorbani *et al*., 2013) and psychology and behavioural studies (e.g. Klingert and Mayer, 2012), as it has the capacity to bridge multiple disciplines.

As with every modelling technique, ABM faces several key challenges that have to be addressed in the forthcoming years. Firstly, there is a need to advance empirical calibration and validation of models (Boero and Squazzoni, 2005; Janssen and Ostrom, 2006) in order to enhance experiment reproducibility and support for policy (Jager and Edmonds, 2015). In a survey by Heath *et al*. (2009), they found the majority of ABMs are not validated both conceptually and operationally. However, more recent literature (Macal, 2016; van Vliet *et al*., 2016) indicates that the situation has changed since 2009, yet only to a certain extent. In respect to calibration, although significant progress being made in empirically rounding ABM mechanisms and agent attributes (Robinson *et al*., 2007; Smajgl *et al*., 2011), ABMs continue to show high subjectivity. Methods to calibrate models include data sources, surveys, semi-structured interviews, participant observation, role-playing games, or laboratory experiments.

The second challenge in ABM is based on linking emergent properties of ABMs to macroscopic patterns of ABMs or other modelling tools. Although there are examples of linking ABMs with other techniques (e.g. with system dynamics models in Miller *et al*. (2014)), this is considered to be a key research frontier for ABM to be addressed in the upcoming years.

Third is upscaling and transferability, referring to scaling-up processes of interactions of a few agents to interactions between many agents (Janssen and Ostrom, 2006). In particular, to explore how social-ecological ABMs can be upscaled to larger geographical areas, considering that an upscaling theory is missing (Arneth *et al*., 2014; Parker *et al*., 2003; Rounsevell *et al*., 2012). For instance, this would enable the coupling of ABMs with models at different spatial scales (Rounsevell *et al*., 2012) and would, thereby, help realize hybrid approaches that couple different models (O'Sullivan *et al*., 2016).

Fourthly, as compared to other modelling techniques (e.g. mathematical modelling), single runs in ABMs do not provide any information on the robustness of the theorems tested, though this can be trivially addressed by analysing output from several runs. Finally, the Agent-Based Land Use Modelling (ABLUM) community highlights specific challenges regarding ABM for the coming years; namely rule definition, i.e. to choose the rules agents use to make decisions, based on the large number of alternatives and the complexity of internal relationships; data acquisition to describe agents' behaviour; and spatial implementation of ABMs.

This thesis addresses several of the above-noted challenges and frontiers through the three ABMs constructed (Chapters 3-5). The way and extent to which each model contributes to help solving these issues is analysed in the Discussion Chapter (Chapter 6). I argue that both the research questions addressed in this thesis (see section 1.2, Chapter 1) and the SES used as case-studies benefit from the dynamic, complexity, agent-heterogeneity and emergent-bottom-up nature of ABMs. Considering that cause and effect are often distant in time and space (Forrester, 1971), the SES used as case-studies in this thesis have complex emergent properties which are essential to understanding the systems' sustainabilities. Thus, the capacity of ABM to model complex systems from the bottom-up, based on interactions between heterogeneous actors, is essential to modelling such SES. Moreover, ABM allows outcomes that occur at one point in time to influence future events – an essential characteristic to model future scenarios and help answering the research questions proposed. Furthermore, very few modelling methods apart from ABM offer the possibility to create spatially-explicit models, as well as hybrid approaches that integrate two or more modelling techniques; this is the case of the model presented in Chapter 5, which integrates BBN in an empirical and spatially-explicit ABM. Besides this, the disaggregated form of computation in ABM can always be aggregated up, while the above-noted modelling techniques cannot always be disaggregated. This is, for instance, an essential characteristic considered for further research in this thesis, based on creating additional versions of the models presented; in particular, to expand the empirical model presented in Chapter 4 to other case-study areas, and to scale-up the spatially-explicit model presented in Chapter 5 from regional to the national level (see Chapter 6 for a description of potential further research for all models).

2.3.3 Agent-Based Modelling to study complex social-ecological systems

SES can be thought of as complex systems comprising feedbacks, sensitivity to initial conditions, stochastic and nonlinear processes, and expressing self-organizing behaviour across scales. Interactions within SES occur among social networks and within communities, along supply chains, and within markets, economies, and ecosystems (Heckbert *et al*., 2010). As both economic and ecological disciplines are concerned with interactions among individuals, both have much to gain from computer modelling tools for complex systems, including ABM. ABMs have been widely used in ecology where they tend to be termed individual-based models (IBM) (Grimm, 1999); they have contributed significantly to ecological theory, including population dynamics, group behaviour and speciation, forestry and fisheries management, conservation planning, and species re-introductions (DeAngelis and, 2005). ABMs have also been widely used in economics, although perhaps to a lesser extent than in ecology (Farmer and Foley, 2009). The field of Agent-Based Computational Economics (ACE) has explored features of economies as complex systems by representing economic agents in computer models as autonomous and interacting decision makers (Tesfatsion and Judd, 2006). An attempt to understand the economy through ABM, and its impacts on the environment, will require the integration of ecological models with models such as financial interactions, real estate, government spending, taxes, business investment, foreign trade and investment, and with consumer behaviour. To achieve this ambitious goal, multidisciplinary collaboration among economists, computer scientists, phycologists and environmental scientists to develop large-scale models would be a first step. The specific topics within ecological economics that could be benefited from such ABM-based collaboration include market dynamics (e.g. Lebaron and Tesfatsion, 2008), changes in consumer attitudes (e.g. Janssen and Jager, 2002), consumption and sustainable behaviour (e.g. Jager *et al*., 2000), natural resource management and land-use change (e.g. Parker *et al*., 2003), common pool resource use (e.g. Schlüter and Pahl-Wostl, 2007), and dynamics of urban systems (e.g. Batty, 2005).

Modelling frameworks for economic and conservation agents

The models built as part of this thesis – presented in Chapters 3-5 – explore the extent to which conservation and economic-development forces drive (un)sustainability, through LUC, in different complex coupled SES. The way in which heterogeneous agents are modelled, i.e. their behaviour and preferences, are described in the corresponding chapters, and determines model results and thesis outcomes. Although each agent has its own particular traits and follows its own decision-making processes (i.e. agent heterogeneity), it was necessary to set a common ground through a robust and theoretically-grounded framework for modelling the two types of agents representing forces driving LUC: economic agents and conservation agents.

Economic agents, in all models, drive resource extraction, production and consumption processes. Representing the main economic agents present in the respective case-studies selected, the economic agents modelled consist of firms extracting and selling resources, households buying and consuming such resources, and banks funding – through credits – resource extraction processes. The main characteristic of these (economic) agents is their profit-seeking behaviour, which enhances (directly or indirectly) a continuous economic growth, through agricultural expansion, regardless of the potential environmental impacts exerted on the environment. Thus, economic agents are self-interested entities and individuals focused on maximizing utility as a consumer and profit as a producer in a competitive market setting. While this context could be related to the *Homo economicus* paradigm (Robbins, 1932) – which argues that humans are rational and narrowly self-interested individuals who pursue their goals optimally – economic agents in this thesis also integrate personal-irrationality, subjectivity, and more complex decision-making processes. Therefore, the main characteristic of the economic agents modelled is their irrational, profit-seeking behaviour – as well as their low environmental awareness; thus, the "only" thing that matters to them is to consume and expand agricultural land to meet demand over goods and gain more profits. Yet, as above-noted, each agent computes its own 'heuristics' and have its own particular characteristics and behaviour – based on specific personal information, such as monetary capital, location, or number of employees.

The approach selected to model economic agents in this thesis aligns with recent criticisms with the conception of *Homo economicus* (e.g. Jones, 2015; Rankin, 2010), which argues that considering market actors as fully rational, self-serving individuals is an overtly simplistic and one-dimensional proposition. The *Homo economicus* framework has been challenged by a wide range of evidence (see Persky, 1995), notably from laboratory economic experiments demonstrating that human decision makers depart from rational and fully informed behaviour. For instance, Heckbert *et al*. (2010) argue that people are at best boundedly rational, typically using heuristics rather than optimization for making decisions, and also show a series of consistent "behavioural anomalies". Thus, people vary in their skills and preferences, value the welfare of others in addition to their own – see Ledyard (1995) – or tend to be risk averse and behave differently when faced with losses or gains.

On the other hand, another relevant type of agent modelled is the one representing conservation forces, i.e. conservation agents. In this regard, government agents in the models do not follow a profit-gaining behaviour, but rather represent those government policies focused on enhancing environmental benefits, e.g. land protection, degraded land restoration. Hence, the goal of government agents in the models is to counterbalance the negative effects exerted on the environment by economic agents (an effect known as 'market failure'). The inclusion in the models of two types of agents (economic and conservation agents), with potentially opposing goals and strategies, sets a suitable context to study the extent to which power (im)balances between economic growth and environmental sustainability enhance SES (un)sustainability. This is because, as previously noted, all the three SES selected as case-studies, including the conceptual system, are characterized for having an environmentally unsustainable economic system, supported and reinforced by different actors and entities, e.g. firms, banks. Therefore, the models presented in this thesis are used to study the extent to which conservation agents, through different plans, strategies and policies, are able to enhance a shift in the mainstream economic growth thinking among these other actors.

Chapter 3:

It's not the '*What*', but the '*How*': Exploring the role of debt in decoupling economic growth from natural resource availability

> *"...If you look at mainstream economics there are three things you will not find in a mainstream economic model - Banks, Debt, and Money.*
>
> *How anybody can think they can analyse capital while leaving out Banks, Debt, and Money is a bit to me like an ornithologist trying to work out how a bird flies whilst ignoring that the bird has wings..."*
>
> – Steve Keen (Australian economist, cited in Southern Energy & Resilience, 2015, p.1)

3.1 Introduction

Humanity has failed to make sufficient progress in solving most environmental challenges, such as climate change, freshwater availability, deforestation, marine fisheries collapses, among others (Ripple *et al*., 2017). This has produced a number of discussions that highlight the impossibility of continuous economic growth within the ecological boundaries of our planet (e.g. Jackson, 2009; Martinez-Alier *et al*., 2010). Therefore, preventing the collapse of the systems that support life on this planet will probably require economic growth to be decoupled from the environmental impact of the economy (Smith *et al*., 2010).

A popular critique of the economic-financial system says that, because banks create money in the form of interest-bearing debt, the system necessarily requires an expanding money supply to pay this interest (Sorrell, 2010). The expanding money supply is argued to enhance an economic growth imperative that forces society to generate an ever-increasing income flow. As a result, more and more debt is accumulated, while more future growth is needed to repay the debt (Daly, 2010). Thus, the cycle continues. This monetary business-as-usual trajectory requires the production of more goods and services (Huber and Robertson, 2000) – along with pollution and resource use – and enhances the probability of system breakdown (Korotayev and Tsirel, 2010).

In this regard, the last financial crisis in 2008 confirmed that the dominant neoclassical models of macroeconomics were seriously flawed (Keen, 2010a). Policy makers, who relied upon models that were not able to predict the actual behaviour of financial markets, were misled, and the credibility of economic theory has been widely called into question (Keen, 2011). Hence, there is a need to develop new economic models that replicate the actual nature of the economy (Keen, 2010a) and transdisciplinary approaches that address the impact of the economy on natural systems (Lang *et al*., 2012; Mauser *et al*., 2013). While there has been much attention on studying the actual nature of both economic and ecological systems independently, the attempts to do so for coupled social-ecological systems (SES) are at an early stage (Fischer *et al*., 2015). SESs are dynamically complex systems composed of people and nature (Redman *et al*., 2004), emphasizing that humans should be seen as a part of, not apart from, nature (Berkes and Folke, 2008). Modelling and exploring coupled SESs is an important step forward, since those economic models not considering environmental implications (e.g. resource availability, pollution) are more likely to show pathways towards false sustainable economic states (Keen, 1995). Yet, the capture of environment constraints, through integration of environmental variables within economic models, could help developing more realistic, long-run scenarios (Giraud *et al*., 2016).

As ecology and economics are concerned with interactions among individuals and entities, both have much to gain from computer modelling tools for complex systems, including Agent-Based Modelling (ABM). ABM simulates systems of autonomous and heterogeneous agents, which interact with each other and their environment, making decisions and changing their actions and the environment as a result of these interactions (Ferber, 1999). ABMs are argued to be helpful for studying complex dynamics in SESs (Balbi and Giupponi, 2010; Filatova *et al*., 2013), as well as gaining insights that support the sustainable management of natural resources (Schulze *et al*., 2017). This paper presents a conceptual ABM that examines the relationship between credit-based economic systems and environmental (un)sustainability, under a complex coupled SES. In particular, the model is used to explore the role of monetary debt in driving the decoupling between economic growth from environmental pressures. For this purpose, the SES modelled integrates a simple environmental-resource system within an ABM inspired by Steve Keen's economic models (2009, 2010a). Keen's work, which was able to reproduce real macroeconomic trends occurring between 1970 and 2010, solved the paradox of how monetary profits can be generated by private actors in credit-based economies. Thus, Keen's explanation shows why the current economic paradigm – based on a continuous and exponential debt-driven economic growth – is strongly supported, and reinforced, by private actors and entities in our society. In particular, Keen was able to simulate how firms increasingly borrow credits (i.e. debt) from banks to finance resource extraction processes and contribute to economic growth, since this provides them with economic profits in the short-run – regardless of their increasing debt burdens. The ABM presented in this paper uses this economic context as a basis – including an environmental system and the economic-environmental feedbacks – to study the relationship between monetary debt and environmental (un)sustainability. The next sections describe the modelling framework in detail, followed by model findings and a discussion on the extent to which monetary debt is a key factor on driving the (un)sustainability of SESs.

3.2 Methodology

3.2.1 Integrating an environmental system into an ABM simulation of Steve Keen's macroeconomic models

The lack of complexity in neoclassical economic models reduces their capacity to describe, in detail, any society ever observed (Moss, 2009). For instance, scholars argue that the mainstream economic models used by some financial entities (e.g. Wall Street) have not been built to understand the complexities of the economic system, but rather to provide tractable results and straight-forward ways to implement policies (Farmer and Foley, 2009). Furthermore, while attempts to model the economic system exist, for instance through system dynamics modelling (e.g. Godley and Lavoie, 2007; Santos, 2007), most economic models only focus on financial processes and do not analyse their impacts on the environment. More specifically, these models have been capable of modelling economic phenomena such as money (Godley and Lavoie, 2012), bounded rationality (Tesfatsion and Judd, 2006) or income distribution (Hein, 2014), yet they are especially weak in regard to ecological variables and to feedback channels between the environment and economy. Thus, the contribution of economic models that explore alternative structures for more sustainable economies, such as “green growth” (OECD, 2011), “steady state” or “degrowth” approaches (Jackson, 2009; Daly, 1991), is rare.

There is a need to understand how the economy affects the environment through complex systems modelling. In this regard, ABM, through the field of Agent-Based Computational Economics (ACE), has explored features of economies as complex systems by representing economic agents as autonomous and interacting decision makers (Tesfatsion and Judd, 2006).Thus, ABM permits the simulation of the economy as a complex system, where human adaptation and learning are taken into account. Furthermore, ABMs have been widely used in ecology, even to a greater extent than economics, under the field of Individual-Based Modelling (IBM) (Grimm, 1999). The capacity shown by ABM to model complex systems, both through ACE and IBM, can be used to simulate SES and explore economic-environmental dynamics in the field of ecological economics. See Chapter 2 (section 2.3.3) for examples of topics within ecological economics that could be benefited from ABM-approaches.

This chapter presents an ABM focused on studying the debt-sustainability relationship within a complex SES. In particular, Steve Keen’s (2009, 2010) economic models is used as a framework in order to build the economic dimension of the ABM – elements from Keen (2009) are also integrated in the mode, mainly Ponzi speculation, which is not included in Keen (2010). Keen’s work is an alternative to traditional economic models that explicitly considers the role of money, debt and banks. The robustness of Keen’s (2010) model lies in a calibration performed against key variables in OECD-economies and the capacity to reproduce real macroeconomic trends and income distributional effects between 1970 and 2010. More specifically, the rationale behind selecting Keen’s work is based on the fact that it was able to explain and justify the paradox of how monetary profits are generated in debt-based economic systems – an issue that economics had failed to provide a satisfactory answer so far (Bruun and Heyn-Johnsen, 2009). In short, Keen’s models show how firms make profits regardless of their dependency on borrowing credits, as well as their increasing debt burdens, which to my knowledge justifies a never-ending economic growth – where unsustainable resource extraction by firms provides profits both in the short- and long-terms. Considering this potentially (environmentally) unsustainable economic framework, an ABM version was built in order to test the impacts of debt-based market economies on the environment, as well as the factors that could enhance the decoupling between economic growth (i.e. GDP) from environmental pressures.

3.2.2 Model description: Overview, Design Concepts and Details (ODD)

The model was built using NetLogo as the modelling software (Wilensky, 1999). Grimm *et al*.'s (2006, 2010) ODD (Overview, Design concepts and Details) model description protocol was used to give an overview of the model. Here the 'Purpose', 'Entities, state variables and scales', and 'Process overview and scheduling' sections of the ODD are included, while the rest of the protocol can be found in Appendix B (pp. 1-27).

Purpose

The purpose of the model is to explore the relationship between debt dynamics and environmental sustainability in credit-based economic systems. More specifically, to study the role of debt in decoupling economic growth (GDP) from environmental pressures, represented by the availability of natural resources.

Entities and state variables

The model consists of agents interacting within three different markets, i.e. credits, goods and labour markets, as well as the environment. The environment consists of a grid of 100 × 100 land parcels (patches), each of them with a biomass (resource) stock. The different types of agents in the model include: firms – which use bank credits to finance production of goods (for which extracting natural resources is needed) that are then sold to households; a commercial bank – which lends credits (loans) to firms under different financial situations; speculators – which bet on the goods (assets) produced by firms, but have no hand in the sale of such goods; and the government – which implements conservation policies to preserve the stock of natural resources and counterbalance the environmental impacts exerted by economic growth.

Figure B–1.1, in Appendix B (p. 2), shows a UML class diagram of the model, specifying and showing the links among model entities and parameters. Table B–1.1, Appendix B (pp. 3-10), shows a description of the parameters modelled for each entity (i.e. agent type), stating whether they are exogenous or constant variables, as well as their initial values.

Process overview and scheduling

The following are the processes that take place every time step in the model. The functions and algorithms computed by these processes are displayed in Table 3.1 – see also the ODD section 'Submodels', in Appendix B (pp. 17-27), for a more detailed description of model functions and processes. Note that some functions are adapted from Keen (2009, 2010a) to our particular modelling context, by disaggregating the equations and algorithms computed by homogeneous entities (in Keen's models) to the heterogeneous nature of ABM. Moreover, new functions with regard to environmental variables are integrated in the ABM, due to Keen's models being purely macroeconomic – where environmental feedbacks are not considered. The model processes include: (i) patches compute biomass stock; (ii) firms extract resources; (iii) households compute demand, movement and energy input/output; (iv) firms compute prices and sales; (v) firms compute labour and finance; (vi) banks compute finance; (vii) firms borrow credits; (viii) firms consider business expansion; (ix) speculators compute speculation; (x) firms and speculators compute credit repayment; and (xi)government computes natural resource conservation policies.

Table 1. Main model functions and the corresponding algorithms.

	function name	acronym	algorithm
[1]	biocapacity	B	$B = R_s \cdot F_y \cdot F_{eq}$
[2]	resource extraction	R_e	$R_e = (D \cdot L \cdot c) - B_r$
[3]	demand	D	$D = H_c/(P \cdot v)$
[4]	investment	K	$K = (AGD_{t-1} \cdot L \cdot F_c \cdot M_c)$
[5]	price	P	$P = (D \cdot P_k) / B_r$
[6]	productivity	p	$p = (F_{c(t)} - F_{c(t-1)}) / L$
[7]	nominal wage	W_n	$W_n = (W_c \cdot F_c)/ L$
[8]	speculation	P_k	$P_k = k(g) \cdot Y$

First, each land parcel (patch) computes one resource stock (R_s), which increases over time following a resource growth function. Related to this, each patch computes its own biocapacity (B) (*function 1*, Table 1), which refers to the capacity of the land to produce useful biomass (i.e. resources with potential to be converted to production goods), and to absorb waste biomass generated by firms (Global Footprint Network, 2018). B varies based on R_s, yield factor (F_y) and equivalence factor (F_{eq}); F_y accounts for differences between countries in productivity of a given land type, while F_{eq} converts a specific land type into a universal unit of biologically productive area (Global Footprint Network, 2018) – note that our model uses the values for forest-land for both F_y and F_{eq}, due to the similarity between the natural resource modelled (in terms of growth-rate and extraction process) and forest-land plantations. Firms extract resources from their current patch location through a resource extracting (R_e) function (*function 2*). The amount of resources extracted by each firm varies with each time step based on households' demand (D) for goods (*function 3*), labour (L) (i.e. workforce), the amount of resources available in firms' biomass reserve (B_r) (which permits firms to cope with periods with excess of demand or lack of resource availability), and a resource conversion factor (c). Firms' resource extraction processes have a monetary cost for them (*function 4*), related to the investment (K) needed, in each time step, to generate enough goods to meet the aggregate household demand (AGD), also considering L, the firm's monetary capital (F_c) and an extraction-demand correction mechanism (M_c). Harvested resources are stored in each firm's reserve, and then sold to households

– after conversion to goods – at a specific price (P) value (*function 5)*; P varies upon D, B_r, and a speculation rate (P_k) (explained below); note that all firms in the model sell the same type of good (modelling different types of good will be subject of a future version of the model). D (household demand) in our model changes based on P, households' monetary capital (H_c), an accelerator effect (v), and distance – note that (v) is related to the GDP, where an increase in GDP enhances (F_c) investment spending in resource extraction. Productivity (p) (*function 6*) states the effectiveness of firms' productive effort, and varies depending upon each firm's profits from one year to the next ($F_{c(t)}$ – $F_{c(t-1)}$) and L. Households work for firms and receive a nominal wage (N_w) following *function 7*.

With regard to the bank, it possesses two different monetary capital stocks – withdrawable capital and bank reserves; while the bank reserve stock holds the monetary capital designated to lend credits to firms, withdrawable capital retains household deposits available for direct withdrawal for consumption of goods. The bank lends credits to firms based on each firm's particular financial situation, and firms have to pay the debt (with interests) back to the bank. The bank also pays deposit interests to households; thus, the bank's net profits vary based on the surplus generated from the difference between household deposits (losses) and credit interests (gains). Credits are used by firms to cover different expenses, i.e. resource extraction processes, wages, investments in improving technological efficiency, and equipment and materials – note that technological efficiency is only applied to the resource extraction processes, i.e. to increase the productivity of extracting resources. Similarly, firms may use credits to fund business expansion, based on creating one new branch/firm in an area with high resource availability. The monetary capital available from the bank for credit lending varies based on the type of economic/banking system modelled (see 'Scenario rationale' below).

Furthermore, speculators also borrow credits from the bank in order to carry out speculative processes (P_k), based on purchasing future derivatives through *function 8*, i.e. instruments to bet on what price the produced good (i.e. asset) will reach by a future date. Speculation increases with further economic growth rate (k_g) and model output (Y), i.e. amount of goods producer per time step. Speculators have no hand in the sale of goods, i.e. they are not the buyer (households) or the seller (firms), yet they are able to affect prices through inflationary and deflationary processes. Both speculators and firms repay credits, with interests, to the bank. Finally, the government monitors the environment, i.e. the availability of natural resource stocks, thus implementing different policies to enhance conservation of resources when the system's natural resource stocks drop below specific thresholds (see 'Scenario rationale' below).

Fig 3.1 shows a UML activity diagram of the model. This shows the links among the above-noted processes and the order in which these processes occur in each time step.

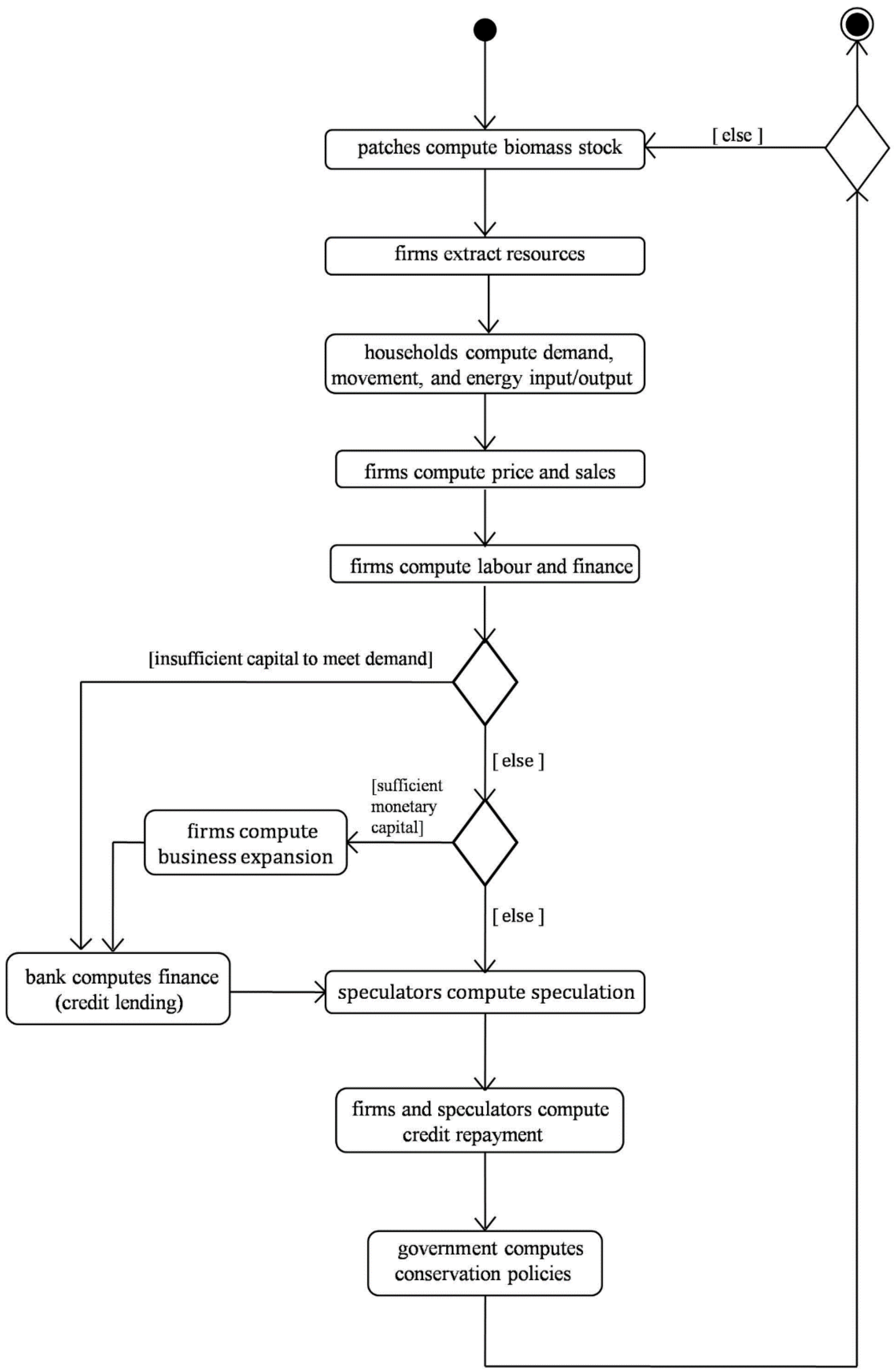

Figure 3.1: UML Activity Diagram. Structure diagram showing the step by step process computed by agents and patches in the model. *Source*: author

3.2.3 Scenario rationale

The model simulates two scenarios; namely fractional-reserve and full-reserve banking systems. The fractional-reserve computes a cash reserve ratio of 0.02 – following the European Union's reserve (European Central Bank, 2011). Cash reserve ratios set the minimum amount of reserves (i.e. the bank's holding of deposits that are not lent out as credits) that must be held by the bank. Thus, 2% is the amount of households' deposits available for withdrawal from banks (i.e. withdrawable capital, for consumption of goods) under fractional systems, while 98% is available solely for credit allocation to firms (i.e. bank reserves). By contrast, the full-reserve banking system computes a cash reserve ratio of 1, where the amount of capital available for credit borrowing is very limited, since the bank must keep 100% of households' deposits available for withdrawal. Due to the gains that the bank makes from the difference between credit interest (gains) and deposit interest (losses) – where the former are normally higher than the latter – the bank, under full-reserve systems, normally allocates more than 0% of capital for credit lending.

Computing both debt-based (i.e. fractional) and non-(or limited) debt-based economic systems allows the comparison of the role of debt in the economy and its impacts on the environment. Moreover, the fractional-reserve system scenario computes various sub-scenarios; these are based on government intervention in the economy through the implementation of conservation policies, which help counterbalancing the negative effects exerted by economic growth on natural resources. Thus, the government in our model enhances natural resource conservation when the total stock of natural resources in the system drops below specific thresholds, provided by the parameter *critical-biomass-stock* (see 'Sensitivity analysis and model calibration' below). More specifically, the policies implemented by the government (i.e. policy options) are focused on (i) forcing firms to decrease investments in technological development to improve production efficiency (i.e. implementation of the precautionary principle); (ii) limiting speculation on assets and speculative artificial markets; (iii) enlarging the protected area network by decreasing the number of patches available for resource extraction; and (iv) forcing firms to restore the land used for resource extraction processes once the natural resources stock is depleted. Note that no government intervention is computed under full reserve system scenarios, due to the very limited impacts exerted on the environment by the economy in this scenario – almost non-existent compared to fractional reserve systems.

3.2.4 Sensitivity analysis and model calibration

An OFAT (One-factor-at-a-time) sensitivity analysis was performed (ten Broeke *et al*., 2016). The sensitivity analysis consisted of observing changes in agents' behaviour, as well as in model outputs, with all except one of the parameters constant. Due to the model being particularly sensitive to changes in the *critical-biomass-stock* parameter, this variable was varied through a series of different values. This parameter states different natural resource threshold values, where the government monitors the total stock of natural resources (biomass) left and implements conservation policies if it drops below predefined values for *critical-biomass-stock*. Thus, the sensitivity analysis performed – see Figure B–1.9, in Appendix B (pp. 19) – shows the extent to which the main environmental (i.e. 'Natural resource stock') and economic (i.e. 'Real GDP growth') indicators are affected under different values of this parameter. Each *critical-biomass-stock* value selected for the analysis was run 100 times, which is considered a reasonable number of runs to generate valid and stable predictions in stochastic simulations (Ritter et al., 2011). The average and standard error values from all the runs regarding the indicators selected are shown in the result figures.

Model calibration followed a comparative analysis between our model's and Keen's (2009, 2010a) results, where the objective was to assess as to whether our model was able to reproduce similar patterns to those from Keen's models. Among the scenarios modelled, the results from the fractional-reserve system (with no government intervention) were used for the calibration process. This is because Keen's models are based on pure debt-based macroeconomic systems, with no full-reserve system included. Furthermore, government intervention in Keen's models do not have the same objective as in our model; where the role of government in his model is to help overcoming an exogenously (to the model) set credit crunch, while our model seeks to explore the endogenous role of conservation governance in preserving natural resources.

Regardless of the conceptual nature of our model, its qualitative behaviour shows matching patterns with regard to those from Keen (2009, 2010a).

3.3 Results

The results analysis compares and identifies qualitative differences in trends among indicators. Fig 3.2 shows the modelling results obtained under non-debt (full-reserve) and debt-based (fractional-reserve) economic systems, the latter also including government intervention through conservation policies for two different *critical-biomass-stock* thresholds (25% and 50%). As previously explained, these values state the maximum stock of natural resources (in per cent values) that need to be left in the system for the government to intervene. The selection of these two values – among a total of twenty – was based on the results obtained from the sensitivity analysis, where 25% and 50% appeared to be critical tipping points with regard to the rest of indicators.

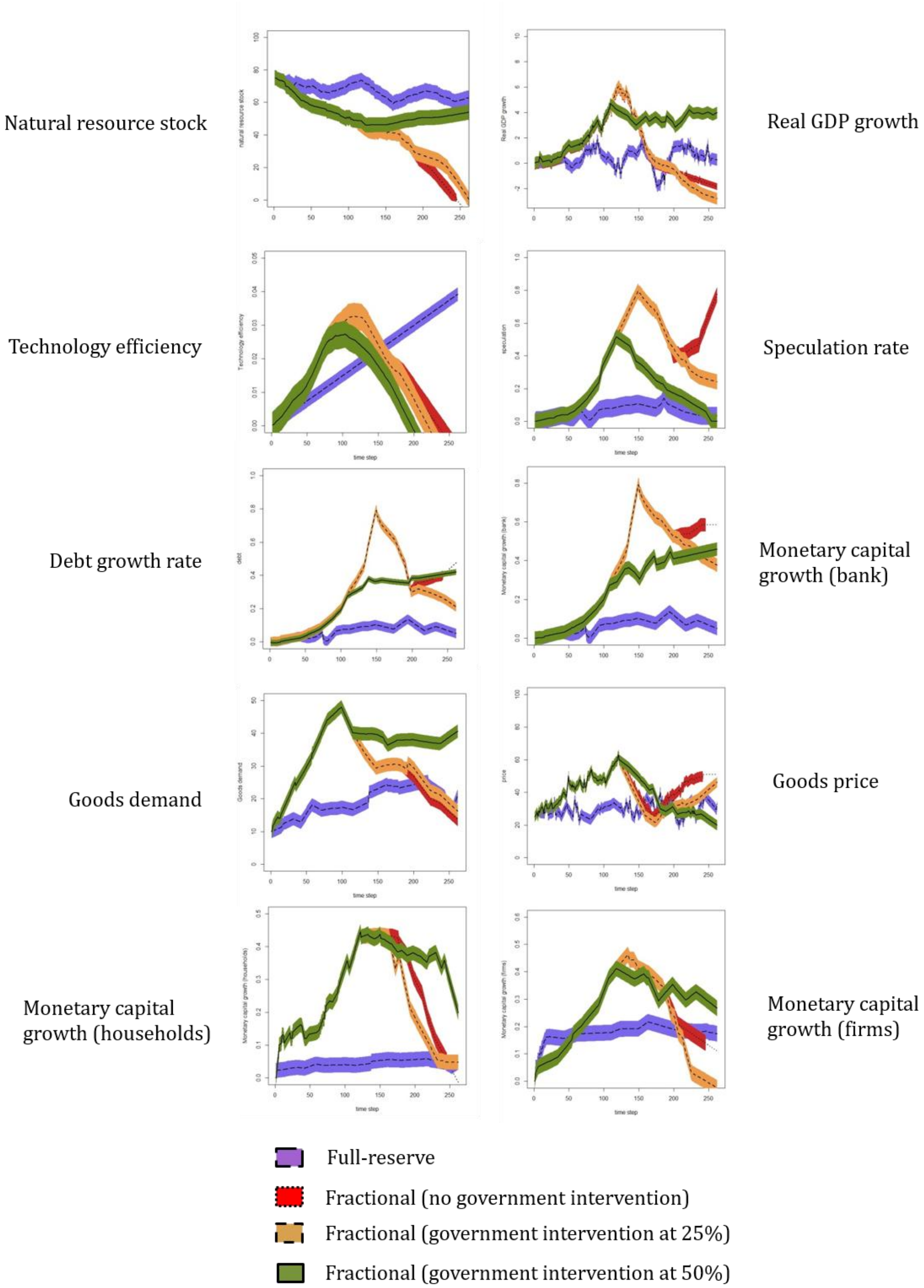

Figure 3.2: Simulation results. Results obtained for the indicators selected under a fractional-reserve system – without government intervention (red dotted line) and with government intervention when the total natural resource stock is at 25% (yellow short-dash) and 50% (green solid line) – and under a full-reserve system (purple long-dash line). Black coloured curves (i.e. dotted, solid, short and long-dashed) show the mean values, whereas coloured bands represent the standard error bars including all the runs computed for each indicator under every scenario.

3.3.1 Debt-based fractional-reserve system (no government intervention)

Under debt-based (fractional) economic systems, with no government intervention, firms are able to cover their daily expenditures, wages and investments for technological development, due to the high availability of natural resources and bank credits. In the processes of selling goods and borrowing (lending) credits, both firms and banks gain profits. Households also benefit from the consequent rise in wages implemented by firms. Overall, this process maintains continuous economic growth, fuelled by loans that drive increasing labour productivity. The increasing level of debt, and the tendency to borrow more when profits increase (in anticipation of a rise in income and the promise of future wealth creation), has no apparent effect on the economy at this point (see the period 0-120 for ‘Real GDP growth’ and ‘Monetary capital’ indicators, red-dotted curve, Fig 3.2). From an environmental perspective, the increasing extraction of resources affects natural resource stocks, thus showing decreasing values during the simulation.

At the same time, the rise in speculation shows that some monetary capital funding economic growth enters the system according to speculative goals, instead of purely production-oriented goals. This is due to the presence of speculator agents, which also borrow credits to gain future profits by trading assets on a rising market. As credit borrowing by speculator agents occurs when prices and GDP increase, this process starts enhancing price inflation and, as a result, further speculation. This reinforcing cycle enhances a growing debt burden that adds no productivity value to the system (see the period 0-120 for ‘Debt growth rate’).

Once the simulation exceeds 100 time steps, price inflation has reached its peak due to speculation, economic growth and the increasing debt burden. As a result, households cannot afford to consume goods anymore, thereby decreasing demand for goods and reducing firms’ monetary capital, which in turn reduces labour – due to the inability of firms to pay for households’ wages. This loss of purchasing power by households enhances a deflationary process, while firms are no longer able to fund investment in technological development for improving production efficiency. Furthermore, price deflation reduces speculation, since the number of speculators in the system is directly correlated with inflationary processes. Thus, most speculators go bankrupt, which reinforces further price deflation. In particular, bankruptcy takes place among those speculators with low monetary capital, who are not willing to borrow further credits. Because most speculators are not able to pay back debt credits to the bank, unpaid debt stocks become the bank's debt. This reduces the capital available from the bank for credit lending, thus creating a domino effect affecting firms and households. From an environmental perspective, the reduction of resource extraction processes benefits natural resource stocks, which show steady state values for the first time during the simulation (see the period 125-175 for 'Natural resources stock' indicator).

Eventually, the drop in prices encourages higher household demand for goods and a period of system stability. However, because this rise is not sufficient to increase firms' monetary capital, GDP values continue to decrease – albeit at a lower rate than under high speculation values. The economy starts to recover slightly, and the rise in prices (>175 time steps) attracts speculator agents again, which enhances debt stocks and further increase in prices – albeit at a lower rate than at the beginning of the simulation. Because natural resources are almost fully depleted from the excessive resource extraction, both firms' income and production of goods are affected, thus reducing the capacity of firms to repay borrowed credits back to the bank. This new context affects banks, firms, speculator agents and households negatively. Eventually, natural resource collapse occurs, thus creating the breakdown of the system and ending the simulation.

3.3.2 Non-debt based full-reserve system

Fig 3.2 also shows the results obtained under a full-reserve system (see purple long-dashed curves in this figure), where the bank is forced to keep 100% of households' deposits available for withdrawal. As previously explained, the amount of capital allocated by the bank for credit lending is not 0% – albeit very low, since the bank still generates money for credit lending from the difference between credit interests (gains) and deposit interests (losses).

Under this scenario, most environmental and economic indicators remain relatively stable over time, compared to those under fractional-reserve systems. Yet, this stability is achieved at low ranges of values regarding ‘Natural resources stock’, ‘Real GDP growth’, ‘Debt growth rate’, ‘Speculation rate’, and ‘Monetary capital growth (firms, households and bank)’, as well as the rest of indicators. Basically, the low allocation of credits (debt) by the bank for both production-oriented (through firms) and speculative (through speculators) goals creates a system with low income and profits, yet also with low environmental impacts. As a result, model results neither show economic nor environmental collapses during the simulation period, since the risk of natural resource depletion, as well as high speculation, debt or inflation rates (which increase the probability of economic collapses) is low.

3.3.3 Government intervention in fractional-reserve systems

The implementation of government policies under a fractional-reserve system was modelled. In our model, conservation governance is used as a process to counterbalance the negative environmental impacts exerted by economic activities. The policies focus on enhancing the sustainability of natural resources only if their total stock in the system drops below two specific thresholds, i.e. 25% and 50% (of the initial stock). Rather than the specific time steps and natural resource threshold values at which conservation policies are implemented (i.e. tipping points), our analysis focuses on the importance of government intervention, as a whole, under potentially unsustainable debt-based economic systems.

Fig 3.2 shows that conservation policies implemented only after ‘Natural resources stock’ drops below 25% of its initial capacity are not able to prevent system collapse (see yellow short-dash curves). In particular, the small amount of natural resources left by then, as well as the high rates of technological development and resource extraction processes, create an unsolvable context for the government in terms of avoiding system collapse. Interestingly, GDP, after government intervention, decreases over time at a higher rate than under fractional-reserve systems with no government intervention. In contrast, conservation policies implemented before the system’s total natural resource stock drops below 50% (green solid curves) are able to enhance natural resource stability over time, with no system collapses during the simulation period.

3.4 Discussion

The model integrates a simple resource-environmental system in an economic-monetary ABM circuit inspired in Keen's models (2009, 2010a). The model reveals the basic dynamics and differences between credit and non-credit economies, where interest-bearing bank debt circulates as money in the former. The model is used to study the impact of debt dynamics on natural resource availability and use, as well as the role of government intervention with regard to counterbalancing the negative impacts exerted by the economic system on natural resources.

It is argued that the results obtained under the credit-based, fractional-reserve economic system represent closely the reality of developed countries today. In particular, the exploration of the viability and sustainability of the system modelled reveals its susceptibility towards instabilities related to monetary debt. Debt is an economic phenomenon that has been widely accepted by neoclassical and the so-called Keynesian approaches of the economy (Keen, 2010b). The current debt-driven monetary system creates the conditions in which continual economic growth – which is the overriding economic objective of most countries – becomes a necessity. The model of debt-fuelled growth requires ever-faster growth rates to allow the repayment of ever-increasing debt (Daly, 2011), and ever-faster growth requires, currently, an ever-increasing production and sale of goods and services, thus increasing the use of resources and emission of pollutants (Huber and Robertson, 2000). Under this context, environmental sustainability is challenged by the depletion of natural resources driven by increasing debt stocks. Thus, the difficulty of avoiding collapse under these conditions may help explaining why monetary debt is a key factor with regard to (un)sustainability outcomes.

Interestingly, our results provide new insights to this debate. Our model shows that the economy does not grow or become unstable due to the debt burden enhanced by the monetary system, or the debt-based nature of the economic system itself – but rather this is the outcome of the *inappropriate use* that firms and speculators make of debt (i.e. bank credits). In fact, results show that non-credit-based systems (i.e. full-reserve) can also create unstable GDP trends over time, even in the absence of debt, thus showing that debt itself is not the main factor driving natural resource depletion. Our fractional-reserve system scenario with early government intervention (i.e. 50%, Fig 3.2) – where the use of debt is production-oriented through technological efficiency and non-speculative processes – shows that the use of debt, rather than the presence of debt in the system, is important to determine the grade of sustainability. Thus, our model shows that the system does not impose a growth imperative *per se*, i.e. the debt-based economic system may not be, by definition, environmentally unsustainable. Rather, agents' behaviour through the use of credits and the system's dynamics show a tendency to increase unsustainability. In short, is not only the "*what*" – the (type of) system – that matters, but the "*how*" (the role of entities and credits in the system, and their relationships with the environment).

The importance of this argument lies in the fact that one of the many criticisms of the monetary system is that the growth imperative is induced by the system itself because society receives less money (the principal of a loan) than that they have to pay back (principal + interest) (Sorrell, 2010); this would induce agents to either monetize and liquidate the natural capital still available as unused resources, or to increase productivity. However, because the total factor productivity (TFP) – which refers to the portion of output not explained by the amount of inputs used in production – only grows at an average of 1.006% in the OECD including energy (Murillo-Zamorano, 2003) – which could be increased to yield the required growth –, economic growth is achieved mainly by using greater stocks of natural resources. Thus, the profit-seeking behaviour of firms and speculative agents drives the inappropriate use of credits (debt), which consequently brings about systemic instability and negative implications for sustainable development.

In our model, the specific uses that firms make of credits are based on (i) processes related to asset speculation and (ii) exponential investments in technological development. The following sections analyse the implications of these elements for (de)coupling economic growth from environmental pressures.

3.4.1 The speed of technological development

In the model, the government implements restrictive credit lending policies that prevent firms from further investing in technological development. Thus, technological efficiency – and with that, the production of goods – grows at a faster rate with no government intervention (see Fig 3.2). Unexpectedly, natural resource collapse, under no government intervention scenarios in fractional-reserve systems, occurs when technology efficiency shows lower values compared to full-reserve systems. It would have been expected that technological development should reach higher values in the former, due to higher investments through credits (see 'Technology efficiency', Fig 3.2). As a result, it is argued that collapses in the model are not specifically driven by the net peak values reached by technology efficiency (i.e. the higher the technological efficiency, the higher the chances for system collapses to occur), but rather by the *speed* (i.e. growth rate) at which technological development takes place. Thus, technology efficiency under full-reserve systems – characterized for firms having less monetary capital available to invest in technological development – reaches a higher long-term net value compared to fractional reserve systems (with no government intervention), yet the speed of reaching this value is higher in the latter. High rates of technological development, therefore, enhance higher resource extraction rates compared to the (normally) lower resource growth rates.

These results show that the rate at which production efficiency increases through technological development, rather than technological development itself, is a key factor for the study of economic-environmental decoupling processes, as well as system collapses. Moreover, such an exponential increase in the rate of production efficiency under fractional-reserve systems (i.e. high debt stocks) encourages a mismatch between government's capacity to implement conservation policies and the promotion of economic growth induced by firms. In our model, the slower pace at which conservation policies are implemented by the government is not sufficient to counterbalance the negative effects exerted on resources by faster technological development rates. In this regard, many OECD governments have been taking steps to adjust their policies to the growing technology and innovation (OECD, 2000), considering that technology efficiency and development have tended to accelerate over the last decades (Modis, 2002). It should be highlighted that economists have usually resorted to technology and innovation as a source of ever-increasing efficiency and economic growth, regardless of the uncertainty and unpredictable nature of technological innovation (Lafforgue, 2008). However, technological progress is, in fact, a discontinuous process, where most significant innovations occur by "fits and starts" (Lafforgue, 2008). The discontinuous nature of technology has the risk of affecting the entire economic system and can lead to far-reaching changes in different social factors (Helpman, 1998), as well as socioeconomic collapse (Blanton and Tainter, 1990; Smil and Diamond, 2005). Moreover, the Jevons Paradox establishes that increases in efficiency of resource use are usually outpaced by the rate at which consumption of those resources increases (Jevons, 1865). Overall, there is a tendency, in our society, to believe in technology despite the lack of support for this proposition; it is either an article of faith or based on statistically flawed extrapolations of historical trends (Brown *et al*., 2011).

In short, our model showed that technological development focused on production efficiency is not a key factor for decoupling GDP from environmental pressures; rather, this is caused by rate (speed) mismatches between both resource extraction and growth rates, as well as between extraction and government policy implementation. It is important to note that technological development can be applied to different fields and, therefore, have different implications for environmental sustainability. In our model, it refers to improving resource extraction efficiency and production processes, thus enhancing the above-noted negative environmental impacts. However, technological development focused on improving waste management (i.e. increase the amount of waste re-used and re-cycled), for instance, would probably be beneficial for the environment. Therefore, it is important to specify and analyse the particular use of technological development at the time of performing sustainability analysis. Under our particular context, a slower, yet constant, increase in technological development, focused on production efficiency (such as that shown under our full-reserve system scenario), could help decoupling GDP from environmental pressures.

3.4.2 Speculation and price volatility

International policy makers and non-governmental actors have become increasingly concerned that the entry of speculators into the system might distort commodity prices by creating excess price volatility (UNCTAD, 2011; Cox, 1976). In our model, the fractional-reserve banking scenario tends to create volatile, artificial and difficult-to-predict speculative markets. Thus, monetary debt is not used by the private sector to increase profits through increasing productivity and, thereby, benefit society (e.g. by enhancing technological efficiency); rather, it is mainly used by speculators to increase their own profits. Our results align with Keen (2009), who states that money funding in the current credit-based economic system occurs according to speculation, instead of production-oriented goals – which enhances the possibility of economic collapses, instability and environmental unsustainability, as shown by our model.

Scholars argue that the rapid increase of commodity derivatives and speculation globally is one financial actor affecting economic trends and, therefore, environmental sustainability (Galaz *et al*., 2015). The commodities for which derivatives are traded are numerous, including agricultural commodities (e.g. coffee, cocoa, soybeans, grains), crude oil and metals. Derivatives for these commodities are being traded in ever-increasing quantities globally due to the entry of new actors, such as large financial investors (e.g. pension funds, sovereign wealth funds) (UNCTAD, 2011). These have a limited interest in the underlying physical commodity, but instead invest in commodity derivatives as a means to diversify their investment portfolios and reduce investment risks (Gorton and Rouwenhorst, 2006). Hyman Minsky's Financial Instability Hypothesis, which has experienced a significant revival since the financial crisis of 2007-2009 (Giraud *et al*., 2016), claims that, in prosperous times – when firms' cash flow rises beyond what is needed to pay off debt – a speculative euphoria develops. Soon thereafter, debts exceed what firms can pay off from their incoming revenues, which in turn produces a financial crisis. A clear example of the impact of speculation on prices, debt and economic instability was observed in the U.S. oil market – see Clark (2006) and PCI (2006) for a detailed description.

Commodity price changes need to be linked to supply-demand processes and the availability of natural resources, rather than speculative processes. Our model shows the extent to which prices and demand processes, under fractional-reserve systems, are highly influenced by economic factors (i.e. the grade of speculation in the system), rather than environmental (i.e. resource availability). Thus, those periods when speculation follows positive increasing trends (see Fig 3.2) show a high disconnection of the economy (represented by the GDP) with regards to the environment (represented by natural resource availability). In contrast, those periods where artificial speculative markets are absent show contexts where economic elements are more coupled to the environment. Under low debt stocks, therefore, the market economy is highly influenced by the state of the environment, i.e. economic growth is aligned with the availability of natural resources, while the opposite is the case in systems with high debt-based speculative processes. It is important to reduce the level of speculation and speculative markets originated in the system, which could help moving towards decoupling GDP from environmental pressure values.

3.4.3 Government responses to environmental unsustainability

Our results show that the economy does not necessarily have to grow or become unstable due to the debt burden encouraged by the monetary system; yet this is the common outcome because of the inappropriate use that firms make of credits, i.e. for speculative and the pace of increasing technology efficiency processes. In the model, this conflict is addressed by implementing government policies focused on enhancing conservation and the more sustainable firm practices when natural resource stock values dropped below specific thresholds. In particular, government policies implemented when the resource stock is lower than 25% of its initial capacity are neither able to enhance a reduction of firms' resource extraction rate nor increase resource growback rates. As previously discussed, the problem lies in the higher speed of technological development compared to the capacity of the government to efficiently respond to environmentally unsustainable practices by the private sector. Related to this, another problem arises based on the difficulty to detect tipping points and predict environmental changes in complex coupled SES (Dawson *et al*., 2010). Complex systems are characterized for having multiple scales, non-linearity and interactive dynamics that are often unpredictable (Axelrod and Cohen, 2001; Holling *et al*., 1998).Therefore, institutions have the difficult task of anticipating the complexity of SES dynamics over multiple temporal and spatial scales to avoid SES collapse, as seen, for example, in common pool resources, such as marine fisheries (Beddington *et al*., 2007; Hardin, 1968). In this regard, system unpredictability is enhanced not only by high technological development rates, but also due to speculation. As previously discussed, prices and demand processes are highly influenced by the grade of speculation in the system under fractional reserve systems, rather than by the availability of natural resources. Hence, high speculative debt-driven economies enhance a disconnection between economic and environmental systems.

The results obtained support the argument that the role of governments should be to invest in preventing market failures through environmental policies that focus on the long-term stability and resilience of the system. For instance, scholars argue that more resilient public institutions and governments are needed in order to be able to adapt to increasingly rapid technological advances (Lebel *et al*., 2006). Thus, a balance is likely needed, where the market still plays an important role in allocating resources efficiently, and the government balances this private perspective with an environmental one (Stiglitz, 2009). The problem here is that, under the current economic paradigm, seeking long-term objectives is penalized by a system focused on short-term gains, generally for the banking and private sector. Thus, increased opportunities should be given to the economic system to invest in both long-term environmental projects and short-term economic ventures, as compared to the situation in which money for loans is only created if it fulfils the profit criteria of private banks (Koslowski, 1995). In real terms, this combination of short- and long-term investments would reduce the ability of individual private actions to constantly expand the money supply and increase the economy's debt burden, as well as halt environmental degradation reinforced by the private sector. Using climate change as an example, and oversimplifying the approaches needed regarding the grade of government intervention into the economy, Nordhaus (2007) argues that limited and gradual government interventions in the economy are necessary. Optimal regulation should reduce long-run growth by only a modest amount. Stern's view (2013) is less optimistic; it calls for more extensive and immediate interventions, and argues that these interventions need to be in place permanently even though they may entail significant economic cost. The more pessimistic answers, such as those coming from degrowth economics (Meadows *et al*., 2004; Jackson and Victor, 2015; Victor and Rosenbluth, 2007), argue that, essentially, all growth needs to come to an end in order to save the planet. We argue that our results stand between Stern and Nordhaus viewpoints: gradual, yet not marginal, and strong interventions under business-as-usual scenarios are needed to prevent the economy from collapsing – not because the current debt-based market economy is, *per se*, unsustainable (as previously discussed), but rather because SES unsustainability is enhanced by agents' and entities' particular behaviour and dynamics.

3.5 Conclusion

The results of our model show that debt-bearing economic systems can result in a complete collapse of both natural and economics systems. Debt is an enabling factor in the exploitation of natural resources for rational individual benefit and short-term gain, hindering long-term environmental and economic sustainability. However, our results show that debt-driven fractional-reserve economic systems do not impose a growth imperative *per se*, i.e. the debt-based system is not by definition unsustainable. Rather, the behaviour of entities and agents, and their decisions and relationships with regards to the environment, show a tendency to increase unsustainability. In the model, the particular uses that firms make of credits are based on (i) speculation, and (ii) exponential investments on technological development. Thus, it is argued that the profit-seeking behaviour of firms and speculative agents drives the inappropriate use of credits (debt), which consequently brings about systemic instabilities and negative implications for sustainable development.

The current version of the model should be considered as a conceptual tool that can be used to theoretically examine the relationship between debt and environmental sustainability. Moreover, the model provides an analysis of the role of the monetary system in the economy and strongly suggests that macro-economic models should incorporate the banking sector if they are to become more relevant. Future versions of the model will include the integration of households as credit borrowers, thus including household speculation and desires. Furthermore, areas for improvement of the model include (1) disaggregating resources into ‘conventional’ (e.g. oil, food) and ‘non-conventional’ (e.g. timber), reflecting higher or lower household consumption dependences on such resources; (2) disaggregating conservation policies; and (3) introducing multiple coupled regions to represent countries with different policies.

Chapter 4:

Exploring sustainable development pathways in debt-based economies: The case for palm oil production in Indonesia

> “*Destroying a tropical rainforest for profit is like burning all the paintings of the Louvre to cook dinner”*
>
> – Edward Osborne Wilson (American biologist, as cited in Friedman, 2009, p. 714)

4.1 Introduction

The model of a debt-based economy may seriously threaten economic development and environmental sustainability (ICSU and ISSC, 2015). Economic growth requires the accumulation of more and more debt, while future growth – fuelled by ever-increasing amounts of energy and resources – is needed to repay the debt (Daly, 2011). And so the cycle continues. Although decoupling economic growth from environmental pressures is at the heart of initiatives such as the Green Economy Initiative of UNEP, frameworks for achieving this goal are still in their infancy (UNEP, 2011). Thus, there is a need to advance the current fragmented and circumstantial evidence and knowledge base regarding the relationship between debt dynamics and environmental sustainability.

The debt-(un)sustainability relationship is highly noticeable in Southeast Asia, where the more than $45 billion in credits lent out between 2010-2017 by overseas banks to companies operating in different sectors (e.g. palm oil, timber, pulp and paper) (Forest & Finance, 2016) are enhancing biodiversity loss (Koh and Wilcove, 2008) and greenhouse gas (GHG) emissions (Pearson *et al*., 2017). Particular attention has to be paid to the palm oil industry in Indonesia, the world’s main debt-driven producer and exporter of palm oil. This industry borrowed more credit facilities than any other sector in the country to fund palm oil production (i.e. USD 9.4 billion), and more than any other palm oil industry in Southeast Asia (Forest & Finance, 2016).

The importance of analysing the relationship between debt and environmental sustainability in Indonesia lie on this country being a focal point for a key trade-off regarding global sustainability trade-off among climate change mitigation, biodiversity conservation and food production (i.e. SES sustainability, as defined in this thesis). First, Indonesia is one of the world's top five GHG emitting countries; this is the main reason why Indonesia has set the goal to reduce its emissions in 26% by 2020 (Paltseva *et al*., 2016). Second, tropical forests in Southeast Asia overlap with four of the world's distinct "biodiversity hotspots", where Indonesia has the highest plant species richness in the world (ICCT, 2016). Finally, Indonesia is the world's biggest producer of crude palm oil (CPO) (USDA-FAS, 2010), and has the objective of near doubling the area for oil palm cultivation by 2020 (UNDP, 2015).

The Government of Indonesia is facing opposing and conflicting goals for 2020 and further – to reduce GHG emissions, halt biodiversity loss and boost food production (Republic of Indonesia, 2016). Can these goals be achieved under a debt-based palm oil industry and economy? The Agent-Based Model (ABM) presented here examines the effects on SES (un)sustainability of power (im)balance scenarios between debt-driven economic forces(i.e. banks and firms) and conservation forces (i.e. governments and public institutions[9]). Thus, the social-ecological system (SES) modelled shows the impact of different future scenarios (2017-2050) on CO_2 emissions, biodiversity loss and CPO production, while other economic and environmental indicators are also considered for model analyses. The short and medium-term governance and policy implications for sustainability in Indonesia are discussed, together with potential long-term 'system rigidity' effects enhanced by power inequalities among the above-noted forces.

[9] As explained in the Methodology, government agents driving conservation refer not only to the Government of Indonesia, but national and international policies and strategies focused on enhancing land protection and degraded land restoration in Indonesia.

4.2 Methodology

4.2.1 Modelling framework

The model was built using NetLogo (Wilensky, 1999)as the ABM construction software. ABM provides a platform to integrate micro-macro processes, use spatiotemporal data, represent human-environment interactions, and combine both qualitative and quantitative approaches (Manson and Evans, 2007). From a theoretical perspective, ABMs are helpful for studying complex dynamics in coupled human-natural systems (Balbi and Giupponi, 2010; Filatova *et al*., 2013), which are characterized by feedback loops, nonlinearity, thresholds, time lags, resilience, among other characteristics (An *et al*., 2014). ABMs have also proven useful in gaining general insights that support the sustainable management of resources through a better understanding of complex SES (Schulze *et al*., 2017). In particular, an increasing number of ABMs are being built within the land-use modelling community, as they offer a way to replace differential equations at high levels (e.g. regional or national scales) with decision rules of entities at a lower level (i.e. individuals or institutions), along with the appropriate environmental feedbacks (Verburg, 2006). Since the earliest published Agent-Based Land-Use Model (ABLUM) (see Lansing and Kremer, 1993), there has been a gradual progression of such models from conceptual land-use frameworks (e.g. Epstein and Axtell, 1996) to more complex empirical representations of SES (Bousquet and Le Page, 2004). Recent reviews and examples of ABLUMs include Filatova *et al*. (2013), Matthews *et al*. (2007), Murray-Rust *et al*. (2011) or Polhill *et al*. (2011).

The empirical model presented here examines the impact of different power (im)balances between economic development forces – driven by monetary debt – and government policy forces – driven by environmental conservation – on SES sustainability in Indonesia. The methodological framework follows the TRACE documentation protocol (Grimm *et al*., 2014), a tool for planning, performing, and documenting good modelling practice. The following sections comprise short characterizations of the TRACE elements corresponding to ‘Problem Formulation’ – including a description of the study area and the scenarios modelled – ‘Model

Description' – using the Overview, Design concepts, and Details (ODD) protocol document (Grimm *et al.*, 2010a) – and 'Data Evaluation' – including model parameterization and calibration processes.

4.2.2 Study area and problem formulation

The study site (Figure 4.1) comprises the provinces of Kalimantan (743,330km^2), Sumatra (473,481 km^2) and Papua (319,036km^2), making a total of 1,525,847km^2. These three provinces, which cover 80% of the total land area in Indonesia, are some of the main producers and exporters of palm oil world-wide. Indonesian annual CPO production, which is expected to be doubled by 2020 (UNDP, 2015), is financially supported by some of the largest commercial banks headquartered in the U.S. (e.g. Bank of America), Europe (e.g. Credit Suisse), Singapore (e.g. DBS), or China (e.g. Industrial and Commercial Bank of China), among others. The palm oil industry in Indonesia received USD 9.4 million in credits between 2010 and 2017 (Forest and Finance, 2016) to cover the upfront costs of producing CPO, i.e. develop land, plant seedlings and build infrastructure (Chain Reaction Research, 2017). Hence, the palm oil industry's reliance on external funding enhances a continuous debt-driven CPO production process that, together with logging, mineral extraction and forest fires, threatens biodiversity and releases significant quantities of carbon to the atmosphere (Carlson *et al.*, 2013). These impacts are particularly important in Indonesia, where the aboveground biomass carrying capacity of some parts is 60% higher than in Amazonia forests (Slik *et al.*, 2010) and being one of the major evolutionary hotspots of biodiversity in Southeast Asia (de Bruyn *et al.*, 2014).

Although various streams of ecological economics offer a biophysical view of the economy, a sound understanding of how key macroeconomic issues – such as global debt dynamics – are entangled with environmental shifts and destructive feedbacks at lower levels is still missing (Klitgaard and Krall, 2012). The model presented in this chapter contributes to this research gap by studying the impact of debt dynamics on synergies and trade-offs among CPO production, biodiversity and CO_2 emissions in Indonesia, upon different economic and conservation forces. Thus, as described in Chapter 1, SES sustainability refers to a context enhancing win-win-win results with regard to these three indicators – where the first two indicators increase and the latter diminishes. The aim of this chapter is, therefore, to produce new knowledge and models that contribute to the 2020 (and further) objective of Indonesia of reducing GHG emissions, halting biodiversity loss and boosting production of agricultural commodities (Republic of Indonesia, 2016). This research objective is addressed by modelling four different future scenarios for the period 2017-2050; namely Business As Usual (BAU), Reduce Biodiversity Loss (RBL), Reduce Carbon Emissions (RCE), and Sustainable Futures (SF). BAU prioritizes exponential economic growth and debt-based CPO production over conservation, whereas RBL, RCE and SF prioritize biodiversity conservation, climate change mitigation and both of them, respectively (see section 4.2.5 'Scenarios' for a detailed description of each scenario).

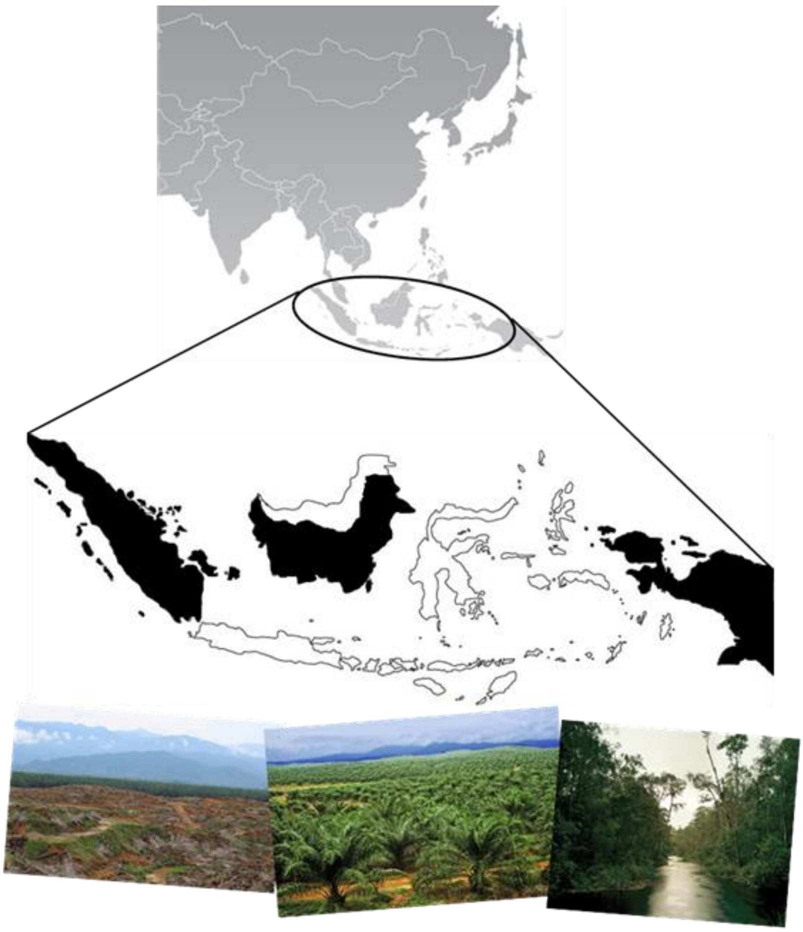

Figure 4.1: Geographic location of the case-study area. Indonesia (top map) and the social-ecological system modelled representing the provinces of Sumatera, Kalimantan and Papua (in black, bottom map). Photographs on the bottom show national examples of degraded land (left), oil palm plantations (centre), and protected primary swamp forest (right). *Source*: author.

4.2.3 Model description

The model description follows the Overview, Design concepts, and Details (ODD) protocol document (Grimm *et al.*, 2010a). The following sections describe the 'Entities, state variables and scales' and 'Process overview and scheduling' elements from the ODD. See Appendix B (pp. 28-50) for a full ODD version describing the model in detail.

Entities, state variables and (spatio-temporal) scales.

The key entities in the model are agents – representing firms, banks and the government – and the environment – consisting of a grid of land-covers. The government agent represents, as a whole, the national and international policies and strategies focused on enhancing land protection and degraded land restoration in Indonesia[10] (e.g. through REDD[11] schemes), whereas the bank represents the overseas financial entities funding CPO production in Indonesia through bank credits. Firms represent the investment groups, i.e. forest-risk groups, financing CPO production in Indonesia.

The model includes a total number of 6,480 patches (i.e. cells, land-covers), which include 14 secondary land-cover types; following Hill *et al.* (2015a),these are aggregated in three primary land-cover types, i.e. 'protected areas', 'semi-natural areas', and 'oil palm plantations'. Simultaneously, 'protected areas' and 'semi-natural areas' are also categorized as 'non-forested' or 'forested', the latter being classified into 'lowland', 'montane', 'heath', 'peat swamp', and 'freshwater swamp'. Figure B–2.1, Appendix B (p. 29), shows a Unified Modelling Language (UML) class diagram describing the model entities and variables in detail. Table B–2.1, Appendix B (pp. 34-38) shows a description of the entities and state variables modelled, their units and data sources.

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[10] Note that the term 'government' is used as an abbreviation (i.e. an agent type name) referring to the public sector, as a whole, driving environmental conservation in the case-study area. Thus, the government agent in the model also includes financial help from international bodies and other developed countries.

[11] REDD, which stands for Reducing Emissions from Deforestation and Forest Degradation, is a United Nations-led program offering incentives for developing countries to preserve and enhance forests.

The model comprises the period 2017-2050; each time step corresponds to one month, thus running the model for 396 monthly time steps (i.e. 33 years). The period 2008-2016 is used for model parameterization and calibration purposes (described below). The time-scale of the model was selected by comparing the modelling outcomes for CPO production and protected area expansion during the period 2008-2016 with historic data for these indicators during the same period. The reason for selecting these indicators to set the time-scale was because these drive the main land-cover changes and outcomes in the model. Thus, analyses of the calibrated results showed a qualitative alignment between both historic data and model outcomes when using a particular time-frame, based on 1 model time step corresponding to 1 month in the real world.

The spatial-scale of the model is considered to be sub-national/national, due to the case-study area comprising a relatively high (i.e. 80%) amount of the total land covered by Indonesia. Furthermore, the patch size selected (235.47ha) under this spatial-scale aligns with the previously selected time-scale (i.e. 1 model time step = 1 month), thus enabling model outcomes to occur at similar rates compared to historic data. For instance, oil palm expansion was responsible for an average of 270,000ha of forest conversion annually from 2000-2011 (Henders *et al*., 2015), making an average of 22,500ha deforested every month, i.e. every time step in the model. The patch size selected (235.47ha) is considered sufficiently large to enable the conversion, at the specific time scale selected, of similar amounts of land into oil palm plantations. The same case applies to protected area expansion and degraded land restoration processes.

Simulation process and overview.

Figure 4.2 shows a UML activity diagram representing the dynamics of the system and the flow from one process to the next one. The following is a list of the model processes taking place every time step, which are described in detail below (model functions and algorithms are shown and described in the ‘Submodels’ section, Appendix B (pp. 39-46)): (i) compute CPO demand; (ii) banks compute credit lending; (iii) firms compute finance; (iv) banks compute credit lending; (v) firms compute resource extraction; (v) firms compute CPO price and sales; (vi) firms compute credit repayment; (vii) firms compute business expansion; (viii) patches compute age and resource extraction; (ix) patches compute indicators; (x) government computes policies.

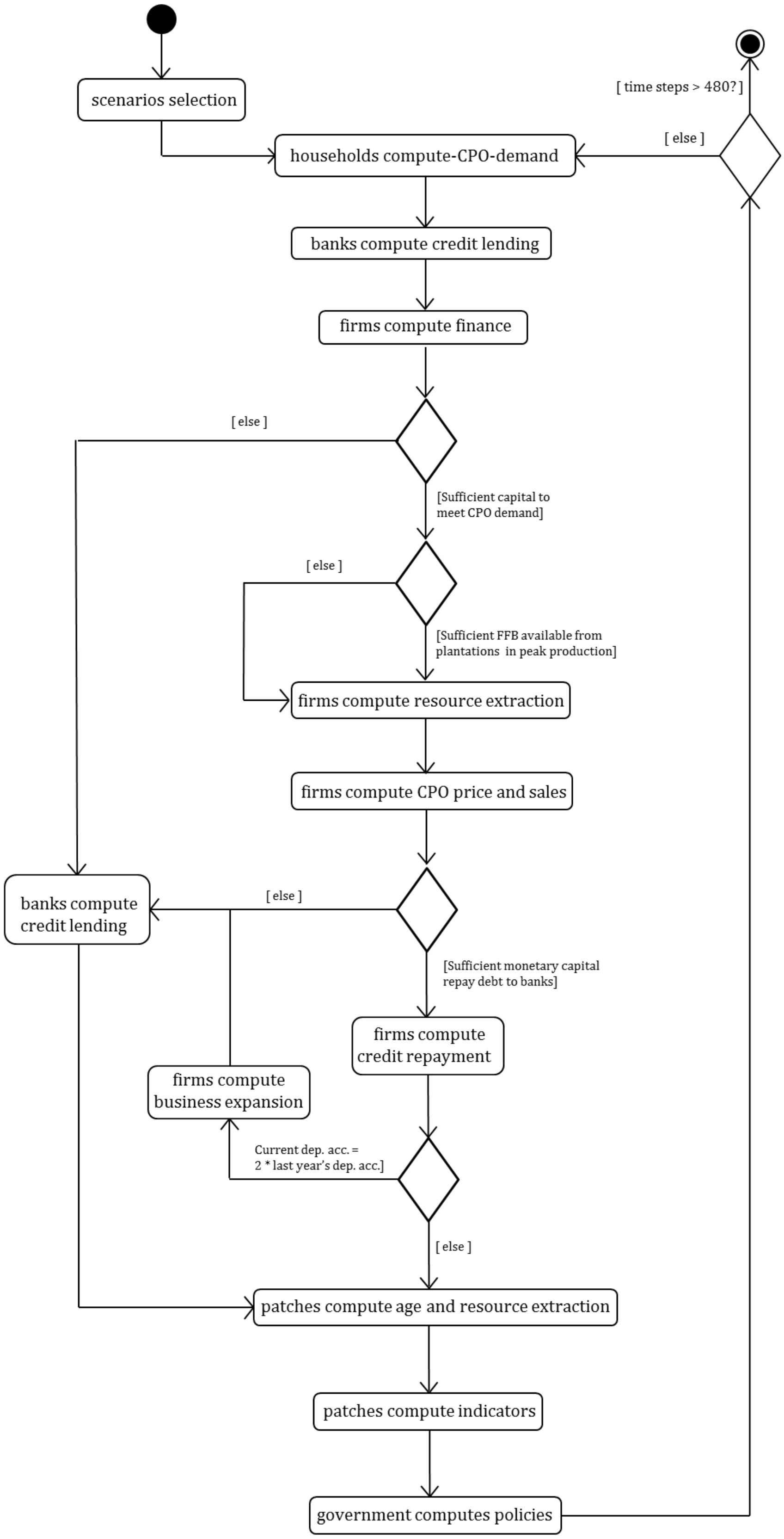

Figure 4.2: UML Activity Diagram. Structure diagram showing the step by step process computed by agents and patches in the model. *Source*: author

The SES modelled is simulated under a credit-based economy, where each scenario(see Table 4.1) sets different rules and limits affecting the dynamics and relationships between agents and the environment. First, CPO demand (exogenously computed) is computed in a monthly basis and distributed among the existing firms based on their current price (explained below); note that CPO demand represents the consumption by households (consumers), which are not agents in the model. Second, firms borrow credits from the bank upfront, i.e. at the beginning of each financial year, in order to cover the direct and indirect operating costs of CPO production in current plantations, including resource extraction, wages for employees and other daily expenditures. Third, firms calculate the final monetary capital needed to cover the expenses for the following month using information on CPO demand; therefore, further credits are borrowed if additional funding to meet the CPO demand is needed.

Firms harvest fresh fruit bunches (FFB) from their owned plantations, which is the fruit produced by oil palm trees from which CPO is obtained. Firms prioritize those plantations where the average tree age ranges between 7 and 18 years – peak production of oil palm trees –since yield gradually decreases after 18 years (Wilmar, 2017). Oil palms begin to produce fruits 30 months after being planted, with commercial harvest commencing six months later. Oil palm plantations with trees older than 25 years – max commercial lifespan –are cut down by the firm owning that land, and are replaced by new plantations with a starting age of 0 from the following month onwards. If firms cannot meet the monthly CPO demand by solely harvesting FFB from peak production plantations, those plantations with trees older than 18 years are harvested until demand is met, followed by those from 3-7 years.

After finishing the monthly harvesting process, each firm sets a price based on a combination of historic information, predicted data and other firms' prices. The firm offering the lowest price is placed at the top of a right-skewed distribution (showing price on the x-axis and demand in the y-axis), thereby being the one prioritized by consumers. When the total CPO demand for that month is met, the CPO selling process stops.

At this point, firms must start paying back their credits, with interest, to the bank. Note that, although firms in the model borrow and repay credits from/to one single bank, this bank agent (the only bank agent in the simulation) represents all financial entities from overseas lending money to oil palm companies in Indonesia. If firms have sufficient monetary capital in their deposit accounts, firms compute the credit repayment corresponding to that month; otherwise firms borrow a credit to cover the debt. Firms also consider expanding their business if their income shows a positive net increase compared to historic data, and their expectations about future profits are positive. In such case, firms borrow a bank credit equal to finance the creation of new oil palm plantations. The identification of potentially suitable sites for new plantations follows Gingold *et al*. (2012) and is a scenario-dependent decision – where firms select areas based on their (CPO) production potential, land-cover availability and conservation potential. Eventually, each firm's monthly income varies upon the profits obtained from CPO sales, and the expenditures regarding the wages allocated to employees, resource extraction process (i.e. materials and equipment, plantation maintenance), technological investments to improve CPO production efficiency, and debt repaid to the bank. While costs associated to debt, wages, resource extraction and material and equipment are mandatory monthly expenditures, firms' investment in technological development is a scenario-dependent decision.

Each oil palm plantation land-cover computes an age function, as well as a stock and a growth function regarding FFB. Furthermore, each land-cover (patch) computes biodiversity and carbon stock algorithms, which changeover time based on the type of land-cover change taking place in that patch and the surrounding matrix of patches. Furthermore, while biodiversity function considers the previous, current and next land-cover changes – both in each patch and in the surrounding ones, thus considering connectivity –, carbon is calculated from losses/gains in above-ground biomass (AGB), which is then converted to carbon and CO_2 respectively. Based on the amount of AGB, each land-cover computes a degradation grade that is used for restoration purposes (see below).

The government (representing national and international public institutions investing in conservation) may intervene in the simulation through different (scenario-dependent) policies. These interventions affect firms' decision-making and, therefore, model outcomes. The government can allocate public funding to firms (i.e. similar to Payments for Ecosystem Services (PES)), as well as create policies, that encourage firms to cover the additional costs for (i) increasing CPO production efficiency on existing plantations by investing in technological development, and (ii) creating new oil palm plantations solely in degraded lands, instead of areas with high biodiversity and carbon stocks – which are, in principle, more profitable. Similarly, the government can invest on (iii) degraded land restoration, and (iv) protected areas (e.g. through funding from international bodies and agreements, such as REDD programmes). The government budget is reduced every time these policies and investments are implemented. The selection of those land-covers to be restored and protected is based on the grade of degradation and the conservation potential, respectively. Finally, the financial opportunity cost of CPO production is calculated at the national level based on the revenue foregone from CPO production as a consequence of restoration and protected area creation.

4.2.4 Scenarios

As previously explained, the model is used to explore four different future scenarios for the period 2017-2050; namely Business As Usual (BAU), Reduce Biodiversity Loss (RBL), Reduce Carbon Emissions (RCE), and Sustainable Futures (SF). Table 4.1 shows a qualitative description of the rationale for each scenario, while Table 4.2 describes the parameters, target values and data sources selected for each scenario.

Table 4.1: Narratives of the scenarios modelled. *Source*: author

Scenario	Description
Business As Usual (BAU)	Rising global demand for vegetable oils drives oil palm plantation expansion in Indonesia, which consequently enhances increasing amounts of borrowed credits from overseas banks to finance CPO production. This process is financially beneficial for both banks and palm oil companies, yet it incurs biodiversity loss and global warming. The Government of Indonesia is more focused on creating jobs and reducing poverty in rural areas through expanding the area of oil palm plantations. This situation is reinforced by the weak environmental governance present in Indonesia, as well as the lack of funding allocated by international organizations for conservation in the country.
Reduce Biodiversity Loss (RBL)	Funding for environmental conservation (mainly from international bodies and developed countries) increases, thus benefiting biodiversity by enlarging the protected area network and restoring moderately degraded forests in Indonesia. Furthermore, biodiversity loss is halted by firms using credits, as well as public funding, to cover the additional costs of creating new plantations in degraded lands – instead of in areas with high biodiversity and carbon stocks – and to increase production efficiency in existing plantations.
Reduce Carbon Emissions (RCE)	The government of Indonesia receives international funding to maximize above-ground biomass accumulation and reduce carbon emissions. Highly degraded forests are restored, due to their high potential to sequestrate carbon. The protected area network is enlarged, yet investments are lower than in RBL since area protection is more focused on halting biodiversity loss. Carbon sequestration is also enhanced by firms using credits and public funding to create plantations in degraded lands (with low carbon stocks) and on increasing productivity in existing cultivations.
Sustainable Futures (SF)	Economically and politically supported by international bodies and other developed countries, the government's goal is to enhance win-win contexts regarding climate change mitigation and improvement of habitat for threatened animal and vegetation species. Restoration of degraded land takes place in both highly and moderately degraded lands, which benefit both biodiversity and carbon conservation. Furthermore, firms use credits and public funding to increase production efficiency in existing cultivations and create plantations in degraded lands.

Table 4.2: Parameters, target values and data sources for each scenario. *Source*: author

	Bank Credits (C) (in USD million)	**Government Budget (GB)**(in USD million)	**Technological Development** (C-funded)	**Oil Palm Plantation Expansion** (C-funded)	**Protected Areas** (GB-funded)	**Restoration** (GB-funded)
Business As Usual **(BAU)**	Between 733.67 and C = 86.126x – 172,457 (‘x’ = model time step)	Maximum of 500	No	From most productive to least productive areas, regardless of biodiversity and carbon stocks.	Increase in 0-3%	No
Reduce Biodiversity Loss **(RBL)**	**RBL_hh** and **RBL_lh** = between 1088.045 and 1442.42. **RBL_hl** and **RBL_ll** = between 733.67 and 1088.045	**RBL_hh** and **RBL_hl** = between 875 and 1250 **RBL_lh** and **RBL_ll**= between 500 and 875	Yes (0 - 7.5%)	From degraded forests, through agricultural lands and secondary forests, to swamp forests.	Increase in 7-10%	Yes – in moderately degraded forests
Reduce Carbon Emissions **(RCE)**	**RCE_hh** and **RCE_lh** = between 1088.045 and 1442.42. **RCE_hl** and **RCE_ll** = between 733.67 and 1088.045	**RCE_hh** and **RCE_hl** = between 875 and 1250 **RCE_lh** and **RCE_ll** = between 500 and 875	Yes (0 - 7.5%)	From degraded peatlands, through agricultural lands and secondary peatlands, to primary forests.	Increase in 3-5%	Yes – in highly degraded lowland forests
Sustainable Futures **(BSF)**	Between 733.67 and C = 86.126x – 172,457	Between 500 and 1250	Yes (7.5 - 15%)	From degraded lands with high production potential, through agricultural lands and secondary forests, to least productive degraded lands.	Increase in 3-7%	Yes – in both highly and moderately degraded forests
Scenario target values sources	Forest and Finance (2016)	Budiharta et al. (2014)	Authors	Budiharta et al. (2014)	Murdiyarso et al. (2011) World Bank (2014)	Budiharta et al. (2014)

The top row of Table 4.2 shows the parameters used to setup each of the four different scenarios, which were selected based on expert opinion. The expert opinion process followed a ‘focus groups’ approach (Kitzinger, 1994; Morgan, 1998; Gill *et al*., 2008), where a group of discussion was organized during one week, in order to provide a deep understanding of the main socio-economic and environmental factors driving SES (un)sustainability in Indonesia. Expertise was sought from five different scientists within a number of fields, including ecology, agricultural sciences, ecological economics, environmental governance, and sustainability science. More specifically, the first day consisted on providing an overview of the topic and the case-study area, as well as the goal of the research work. The second day consisted on asking open-ended questions to the experts, as well as collecting information and data sources from them, about the main SES (un)sustainability issues occurring in Indonesia; these focused on the current economic-conservation dichotomy present in Indonesia, the highly dependency of the palm oil industry on external financial institutions, and the potential conservation policies being implemented to counterbalance the negative environmental impacts exerted by such scenario. The last three days consisted on specific and deeper discussions, including ending questions to experts, on what particular scenarios (including target values) and key factors/variables, under a SES perspective, should be integrated in the model with regard to analysing the current environmental (un)sustainability context in Indonesia. The resulting scenarios and factors selected are displayed on the top row of Table 4.2.

The values of these parameters change from scenario to scenario (left column), while the last row shows the sources from which the different scenario values were obtained. In particular, ‘Bank Credits (C)’ and ‘Government Budget (GB)’on the left are the two economic parameters financing the rest of parameters from the top row. The ranges of values for all parameters include all the possible values that agents can compute in each modelling time step. Note that the position of ‘l’ and ‘h’ letters under RBL and RCE scenario names refer to ‘low’ and ‘high’ Government Budget (GB) (if placed in first position) and Credits (C) (if placed in second position), respectively; for instance, RBL_lh refers to ‘Reduce Biodiversity Loss’ scenario with ‘low’ GB and ‘high’ C (see the Results section for a more detailed explanation). Technological development shows the maximum and minimum (per cent) monetary value that each firm can invest, in a monthly basis, in increasing production efficiency in existing cultivations. Both 'Oil Palm Plantation Expansion' and 'Restoration' describe the type of land cover prioritized, in descending order, regarding the creation of new oil palm plantations and for restoring degraded areas, respectively. Note that the prioritization of moderately degraded forests for restoration under RBL is based on these having higher potential for biodiversity than carbon sequestration, since non-native commercial tree species could be replaced with highly diverse native species. Similarly, highly degraded forests have higher potential for carbon sequestration than for biodiversity conservation. Protected Areas shows the maximum and minimum amount of land (in per cent values) to be protected during the entire simulation period (going from higher to lower conservation potential areas); RBL values are higher than RCE due to area protection being a strategy more focused on improving biodiversity outcomes rather than climate change mitigation (Murdiyarso et al., 2011).

4.2.5 Data evaluation and run setup summary

The updated TRACE format (Grimm *et al*., 2014; TRACE, 2014) merges both model parameterization and calibration processes under 'data evaluation'. Augusiak *et al*. (2014) define data evaluation as the assessment of the quality of the empirical (e.g. published) and qualitative (e.g. expert knowledge) data used to parameterize the model via calibration.

The empirical nature of the model is represented by the different data sources and historic data integrated within the model (see Table B–2.1, Appendix B, pp. 34-38). To be highlighted is the use of historic banking data on debt, which drives bank and firm agents' decision-making processes – obtained from the dataset Forest and Finance (2016). This dataset shows up-to-date information regarding the amount of credits lent by international banks to different industrial sectors in Southeast Asia, in order to fund the production of different goods and services, e.g. palm oil, timber, cotton. The dataset includes the name of the banks, types of industries, name of firms, type of credit facilities, amount allocated per year, among other information. Other empirical data integrated in the model includes the area covered by each land cover, oil palm trees' growth rate, CPO prices, carbon sequestration rates, among others.

Model parameterization focuses on exploring model parameter values, including a list of all the parameters and values, the data sources, and how the parameter values were obtained (Railsback and Grimm, 2011). Table 4.1 describes the narratives of the scenarios modelled, while Table 4.2 shows the parameters, values and data sources regarding each scenario. Table B–2.1 (Appendix B, pp. 34-38) shows the model entities and their state variables, as well as the parameter types, selected values/units and source of datasets.

For the purpose of model calibration, both historic and literature data sources for the period 2008-2016 were used; the aim was to determine the values for Credits (C) and Government Budget (GB) (left part, top row, Table 4.2) parameters, as well as the number of firm agents in the model. The number of government and bank agents was not calibrated as only one of each type was modelled. The selection of these three parameters for model calibration is due to these being the main drivers of model outcomes. More specifically, the importance of Credits (C) parameter lies on its direct effect on CPO production, biodiversity and CO_2 emissions, as well as other sustainability indicators, through technological development and oil palm plantation expansion; similarly, GB drives land conversion of protected and restored areas, thus affecting biodiversity and CO_2 emission indicators. Moreover, the importance of calibrating C and GB lies on the fact that these constitute the main two parameters used to set the Power Imbalance values (see Results).

The full model calibration process, including calibration results, is described in detail in Appendix B (pp. 47-50). In brief, a direct calibration was performed for C, while an inverse calibration was performed for GB and number of firm agents; thus, while historic data available regarding credits borrowed by palm oil companies (i.e. C) permitted us to fit historic data with model outcomes, the lack of data for government budget (i.e. GB) and the number of firms forced us to use alternative historic data from other indicators in order to fit such data to model outcomes. In particular, the expansion/contraction of the protected area network and CPO production were used to calibrate GB and firm agents, respectively.

The results produced are analysed below, and were obtained by computing each scenario 20 times, thus making a total of 80 runs – where each simulation is run for 396 time steps (33 years, 2017-2050); note that the scenarios were calibrated prior to computing the simulations. The average and standard error values from all the runs are shown in the result figures.

4.3 Results

The results obtained (see Figures 4.3, 4.4 and 4.5) were analysed to compare and identify differences in trends among the indicators selected, within and between the scenarios modelled. The qualitative analysis performed followed the main interest of exploring the magnitude of differences among indicators and scenarios; that is, to examine the positive or negative trends for each indicator, as well as the differences (in trends) among indicators and scenarios – rather than quantitatively analyse the statistical significance of the results. This decision was supported by the considerably high differences in the results obtained for each indicator and scenario modelled (as shown by Figures 4.3, 4.4 and 4.5), where a statistical analysis would not contribute with regard to improve the understanding of the results.

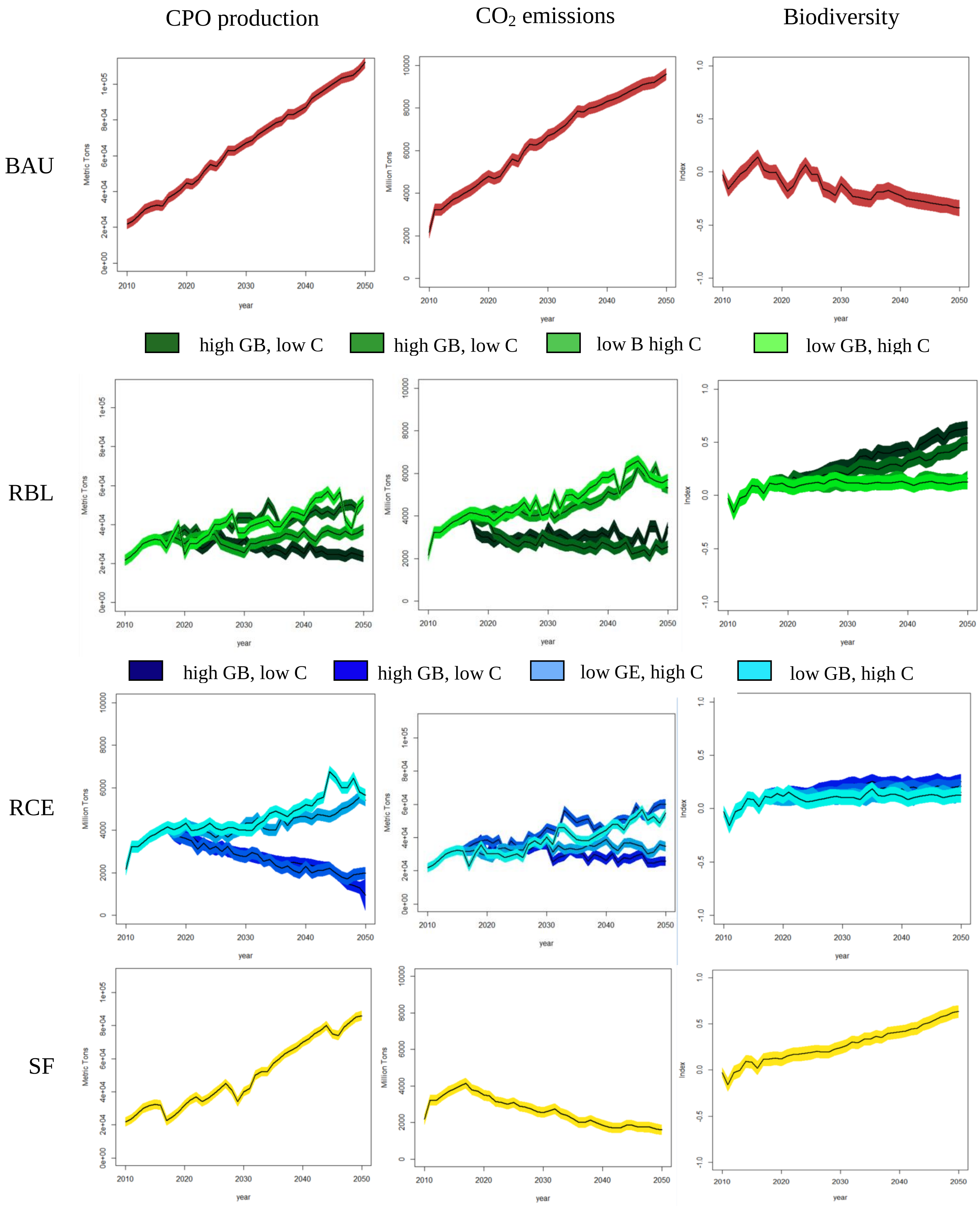

Figure 4.3: Results obtained for SES sustainability indicators. Indicators include crude palm oil (CPO) production (metric tons), CO_2 emissions (metric tons) and biodiversity (index). RBL and RCE scenarios are divided in four different sub-scenarios: dark green (or blue) coloured curves refer to scenarios with strong conservation (high GE) and weak economic (low C) forces (this also applied for Figures 4.4 and 4.5).

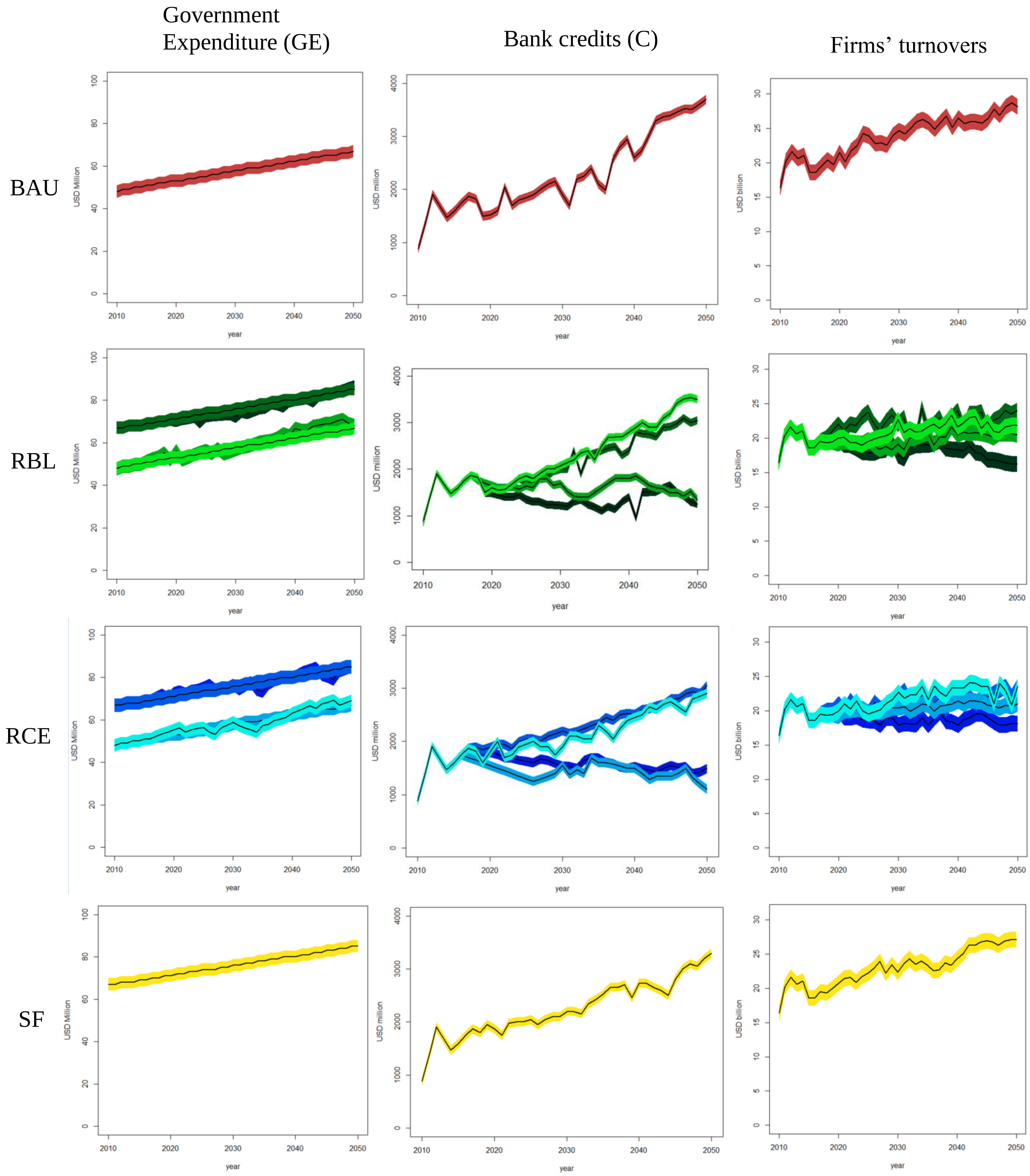

Figure 4.4: Results obtained for monetary-economic indicators. Indicators include bank credits (corporate loans and revolving credit facilities, USD million), government expenditure (USD million) and oil palm firms' turnovers (USD billion). See Figure 4.4 for the legend regarding RBL and RCE sub-scenarios (i.e. colour gradient legend). Note that, while GE results refer to the actual (final) expenditure of monetary capital by the government for conservation purposes, GB categories (high or low) represent the (initial) amount of capital (budget) available for that purpose.

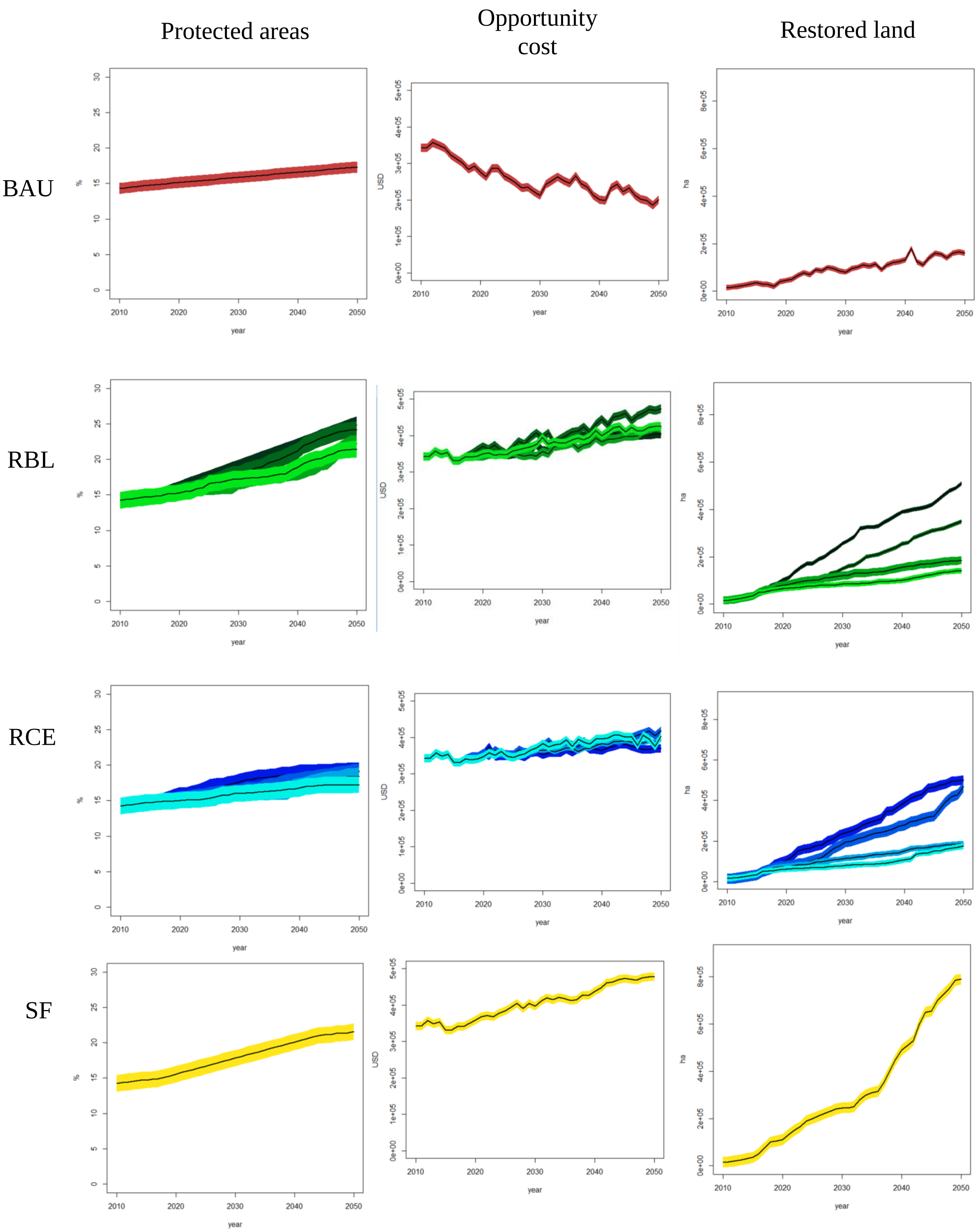

Figure 4.5: Results obtained for environmental-economic indicators. Indicators include protected areas (percent values), restored land (ha) and opportunity cost for the oil palm industry of increasing protected areas and restoring land (in USD). See Figure 4.4 for the legend regarding RBL and RCE sub-scenarios (i.e. colour gradient legend).

4.3.1 Business As Usual (BAU)

The first row in Figures 4.3, 4.4 and 4.5 shows the results obtained under BAU. This scenario shows the highest values for most economic indicators, while environmental outcomes show mainly negative trends. It is argued that this is due to protection forces in Indonesia not being sufficiently strong to halt the economic forces driving land clearing for CPO production (analysed in the Discussion section). Thus, oil palm firms require a continuous flow of bank credits to expand oil palm plantations – normally into areas with high biodiversity (e.g. undisturbed upland forests) and high carbon stocks (e.g. swamp forests). As a result, the number of credits borrowed and CPO production increase over time, while the opportunity cost of not converting land into oil palm plantations continues to decrease. Concurrently, biodiversity loss and CO_2 emissions are reinforced, also due to the reduced government budg*et al*located for conservation purposes– which still shows a steady increase over time.

4.3.2 Reducing Biodiversity Loss (RBL) and Reducing Carbon Emission (RCE)

The second and third rows in Figures 4.3, 4.4 and 4.5 show the results obtained for RBL and RCE scenarios, respectively. As indicated in the respective footnotes, each RBL and RCE scenario is divided in four different sub-scenarios, with varying values regarding two variables: C (bank credits) and GB (government budget). In particular, four different sub-scenarios are considered, namely high GB and low C, high GB and C, low GB and C, and low GB and high C. These refer to the amount of monetary capital initially (i.e. at the beginning of the simulation) available for driving conservation and for land clearing processes (as credits available for borrowing), respectively. The specific values for each case are shown in Table 4.2.

RBL and RCE scenarios show similar trends for most indicators, which, as per SF scenario (see below), minimize land requirements by intensifying CPO production. Some monetary-economic indicators (credits borrowed by firms and firms' turnovers), as well as some environmental indicators (CPO production)show more negative results than those under BAU, due to economic forces driving land clearing for oil palm production being weaker than conservation forces. Under RBL, strict enforcement of forest protection enhances the creation of new protected areas, land restoration and the creation of new policies that force firms to decrease the number of new plantations in areas with high biodiversity. Biodiversity, therefore, increases with higher GB values; the same context occurs for CO_2 emissions, where more sustainable results are obtained under scenarios with high GB values. The main difference between RBL and RCE in terms of biodiversity and CO_2 emissions is based on the type of forests restored: while moderately degraded forest is least favoured for restoration under RCE, highly degraded forest is least favoured under RBL, thus enhancing higher biodiversity values under RBL and lower CO_2 emissions under RCE (explained in Table 4.1).

4.3.3 Sustainable Futures (SF)

The fourth row in Figures 4.3, 4.4 and 4.5 shows the results obtained under SF scenario. This is the only scenario showing synergies between CPO production, CO_2 emissions and biodiversity, as well as relatively positive results for the rest of indicators. Interestingly, this is achieved under the same credit-based economic system as the one modelled under the BAU scenario, where the number of credits borrowed by firms increases over time. These results are obtained due to the combination of the following factors: (i) the use of technology by firms to increase production efficiency in existing cultivations, which significantly minimizes land requirements for CPO production; (ii) the creation of new plantations solely in degraded lands, thus avoiding plantation expansion into areas with high biodiversity and carbon stocks; (iii) the increase in the amount of degraded land restored; and (iv) the increase in the number and extent of protected areas. As analysed in the Discussion section, implementing these policies have economic implications to be covered by both firms and the government.

4.3.4 Environmental impacts of Power Imbalances between banks and government.

Results shown in Figure 4.6 (below) allow us to explore the extent to which biodiversity and CO_2 emission values vary under different Power Imbalance contexts between banks (represented by credit allocation to firms, i.e. C) and the government (represented by the budget allocated for conservation from both national and international public entities, i.e. GB). Power Imbalance values (i.e. *x-axis*) are calculated through a simple C/GB function that states the proportion of total credits available (C) to the government budget for conservation (GB). Power Imbalance values range from high (on the right-hand side of the x-axis in both heatmaps) – where the amount of bank credits available for CPO production (C) is considerably higher than the government budget allocated for conservation (GB) – to low Power Imbalance values (on the left-hand side of the *x-axis* in both heatmaps) – where GB and C show similar values, or even GB being higher than C. Note that placing C as a numerator and GB as a denominator in the C/GB function favours economic development over conservation. This decision was made following Hill *et al*. (2015a), who argue that the current economic forces driving land clearing for production in tropical countries are stronger than conservation forces driving land protection and restoration. Hence, high GB values in the model can only but equilibrate the power distribution between banks (economic forces) and conservation governance (conservation forces), yet never shift it towards favouring conservation over economic – due to the current BAU reality analysed in Hill *et al*. (2015a).

Thus, the Power Imbalance values between economic and conservation forces are plotted against the different scenarios (i.e. BAU, RBL, RCE and SF) placed in the *y-axis*. As previously explained, both RBL and RCE scenarios are divided in 4 different sub-scenarios each, showing the amount of monetary capital initially available for driving conservation (GB) and for land clearing processes (C), respectively. The following list disaggregates the abbreviations shown in the *y-axis* for the RCE scenario, where 'h' refers to 'high' and 'l' to 'low' (the same case applies to RBL). See Table 4.2 for the values referring to high and low:

- RCE_ll: RBL scenario with low (l) GB and low (l) C.
- RCE_lh: RBL scenario with low (l) GB and high (h) C.
- RCE_hl: RBL scenario with high (h) GB and low (l) C.
- RCE_hh: RBL scenario with high (h) GB and high (h) C.

In short, Figure 4.6 below shows the impact on biodiversity and CO_2 emissions of the scenarios modelled based on different power imbalances between economic and conservation forces. Note that each scenario (*y-axis*) does not only consist on different

GB and C values (which are used to calculate the Power Imbalance values shown in the *x-axis* through the C/GB function), but also include other different processes and properties characteristic of each scenario (i.e. different rates of technological efficiency selected for CPO production by firms, potential areas selected for restoring degraded lands).Focusing on the results shown by Figure 4.6, one can see the negative impacts that high Power Imbalance values, under the BAU scenario, exert on both biodiversity and CO_2 emissions. In contrast, the SF scenario shows considerably high values for both indicators. The different sub-scenarios represented by RBL and RCE show varying results regarding both indicators, including trade-offs among sub-scenarios, with predominantly better results obtained for biodiversity under RBL and CO_2 under RCE.

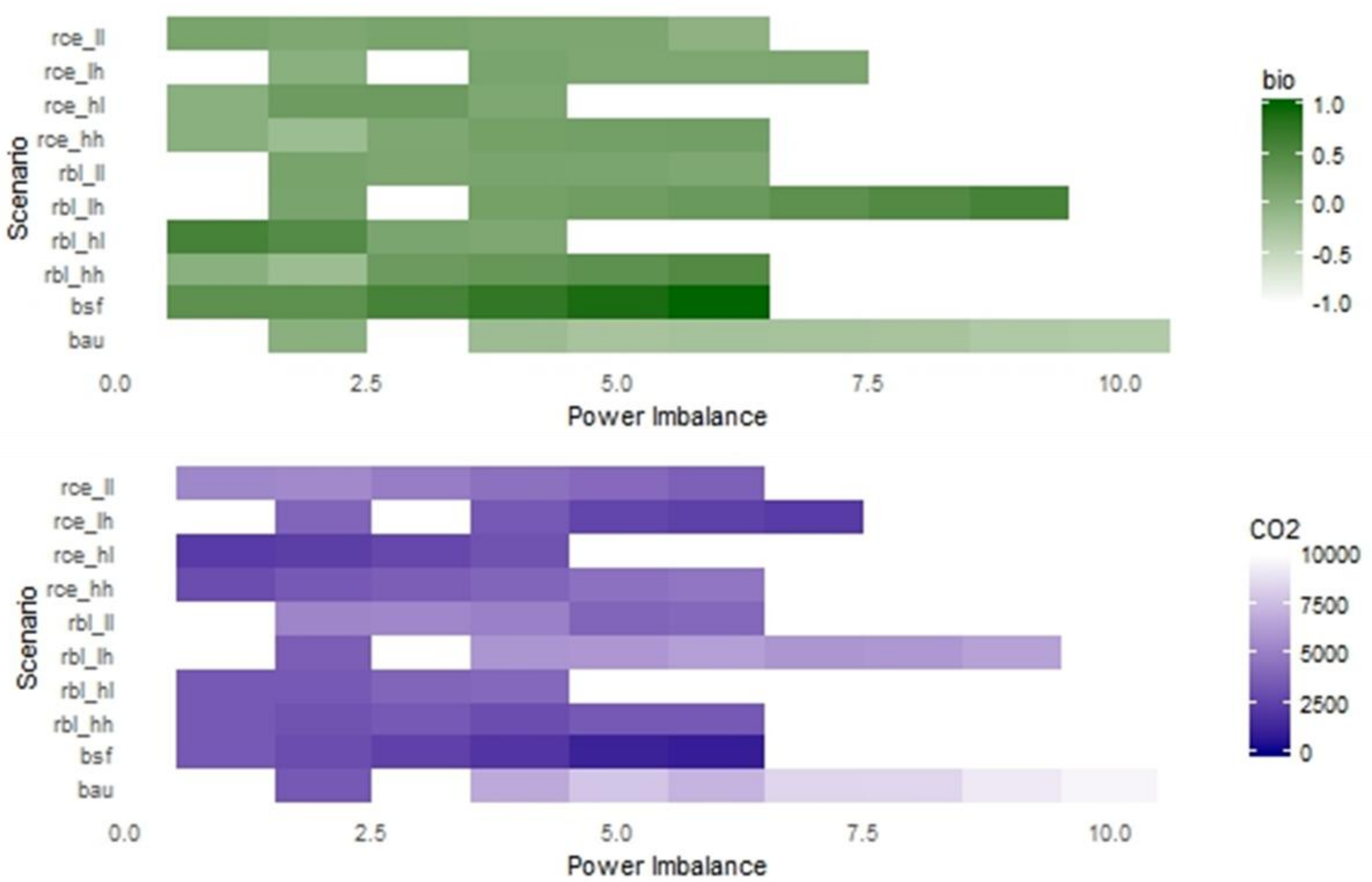

Figure 4.6: Impact of different power (im)balance rates between economic development and conservation on biodiversity (top heatmap) and CO_2 emissions (in millions of tons, bottom heatmap). See page 99 above for an explanation of the abbreviations in the y-axis. Dark coloured cells represent high values, while low values are represented by light coloured cells. Cells in blank show those scenarios with neither biodiversity nor CO_2emission values computed for certain Power Imbalance values. *Source*: author

4.4 Discussion

4.4.1 Analysing the relationship between conservation forces, economic powers and SES sustainability in Indonesia

Oil palm development, forest conservation and climate change mitigation are strategies that are, at first glance, in opposition to one another (UNEP, 2011). The model addressed the inherent conflicts by simulating a SES that represents the dynamics and relationships between financial institutions, palm oil industry, conservation forces and the environment in Indonesia.

Considering that the same credit-based economy is modelled under both BAU and SF scenarios, the main factor driving SES (un)sustainability among these scenarios is the (in)appropriate use that oil palm companies make of bank credits, rather than the amount of credits borrowed or the debt-based nature of the economic system itself. Thus, the problem causing unsustainable outcomes in the model is not an economic system that creates a high dependency of different actors on debt and banks (i.e. the '*what*'), but the purpose for which credit facilities are borrowed and allocated (the '*how*'). In fact, the SF scenario (Figure 4.6) shows synergies between debt (i.e. credits borrowed), high biodiversity and low CO_2 emissions, showing that an appropriate use of bank credits may be beneficial for SES sustainability. These synergies are achieved under moderate and high Power Imbalance values (see darker SF cells on the right-side of both heatmaps), i.e. when the power/influence of banks is higher than conservation forces. Results, therefore, align with recent research arguing that the current utilization of credit facilities by firms – focused on covering daily operating costs and expanding oil palm plantations into areas with high biodiversity and carbon stocks – is a key problem for sustainability in Indonesia (Alwarritzi *et al.*, 2015). In this regard, scholars argue that using bank credits to increase production efficiency in existing oil palm cultivations could be sufficient to meet the growing world demand for Indonesian CPO, while help conserving the highly valuable habitats in terms of biodiversity and carbon sequestration (Fairhurst, 2009). For instance, Fairhurst (2009) shows that yield improvements in existing cultivation alone could potentially avoid the need to expand into 1.6 million hectares of forest. Simultaneously, other authors argue that yield improvements alone will not be able to provide sufficient CPO to meet the growing world demand over the next thirty years (Fairhurst and McLaughlin, 2009). Rather, planting oil palms in degraded land is suggested as an alternative solution to help meeting global CPO demand, while avoiding the release of excessive CO_2 to the atmosphere and further biodiversity loss (Koh and Ghazoul, 2010).

The problem with alternative solutions focused on enhancing sustainable CPO production is that debt-dependent palm oil companies, likewise firms under the BAU scenario, will unlikely use credit facilities to finance less profitable, financially riskier 'innovative' CPO production processes – i.e. create new plantations in degraded lands and invest in technological efficiency. Currently, traditional oil palm cultivation in biologically rich areas provides firms with higher short-term profits, due to the low price of land in these areas. In fact, 'innovating' palm oil companies (i.e. firms implementing the above-noted sustainable strategies) would probably be under-cut on international markets by traditional palm oil producers from other countries (i.e. BAU firms). Therefore, there is a need to financially support oil palm companies in order to shift their BAU paradigm for a more sustainable one. In the model presented, government funding is allocated to oil palm companies under the SF scenario (together with the usual credit facilities), which helps firms covering the additional costs of increasing technological efficiency and establishing new plantations in degraded lands. Thus, it is argued that new financial mechanisms could help firms covering the currently higher costs of adopting more sustainable practices (see Ruysschaert *et al.*, 2011); until technological development or the market itself – due to land scarcity – starts inherently supporting more sustainable pathways for CPO production, i.e. due to a reduction of prices and operational costs related to more sustainable practices.

In this regard, Payments for Ecosystem Services (PES) (Farley and Costanza, 2010; Wunder, 2005) could be considered a potential temporary solution within the transition towards more sustainable scenarios. PES seek to ascribe monetary value to ES (Bellver-Domingo *et al.*, 2016), for instance through international schemes, including the REDD and REDD+ programs. REDD(+) offers incentives for developing countries to preserve and enhance forests, thus offsetting the growth in global GHG emissions (and biodiversity loss) (Angelsen, 2008).The model presented here shows that, if sufficient funding is allocated to firms for enhancing sustainable CPO production, short and medium-term synergies among the sustainability indicators explored could be reinforced. In fact, results show that this could take place without having to replace the credit-based production system nor reducing the current power of banks. Thus, there is a need to increase the socio-political and financial support from international bodies to Indonesian oil palm companies (e.g. PES schemes) to enhance the delivery of multiple beneficial ES and diminish the environmental impacts of traditional investments of credit facilities. As an example, Indonesia signed a US$1 billion deal with Norway in 2010, under the REDD framework, aimed at reducing deforestation in Indonesia as a follow up of the United Nations Climate Change Conference held in Cancun, Mexico (Lang, 2010a). Furthermore, it is expected that 200 billion Euros will be transferred world-wide through PES schemes by 2020 (GIZ, 2016). The problem here is that funding for development is usually much higher than for conservation (Hill, 2015c). For instance, the leaders of the G20 nations gave a huge boost to the power of development regimes by promising to invest 60-70 trillion U.S. dollars on new infrastructure projects by the year 2030 (Hill, 2015c). Therefore, if the objective is to be able to compete with the agricultural sector, investments for PES and other financial mechanisms need to increase (Butler *et al*., 2009).

Besides enhancing a more sustainable use of credits by firms through PES, results obtained under low Power Imbalance values in the SF scenario (see darker coloured cells under RBL_hl and RCE_hl scenarios, Figure 4.6) show the need to enhance conservation forces in terms of increasing protected areas and restore degraded land. Under these scenarios, the number of credits available for borrowing (which, as above-noted, can help enhancing sustainability) is low, yet this is compensated by an increase in the power of governance forces driving land protection and restoration. Thus, there is also a need to enlarge the current protected area network and restore part of the 46.7 million hectares of degraded land currently present in Indonesia. The results obtained align with recent research stating that the creation of protected areas in Indonesian forests is a less effective way than restoring degraded land with regards to halting deforestation and biodiversity loss (Symes *et al*., 2015); yet, they are not be mutually exclusive. In this regard, Murdiyarso *et al*. (2011) criticized the Indonesian Government for giving a higher importance to land protection than restoration through the above-noted REDD+ agreement with Norway. Thus, findings from the model presented support the proposal for the inclusion of part of the 400,000 hectares of highly degraded lowland forest into the moratorium.

Overall, results from the SF scenario show evidence that, with various adjustments, a compromise solution regarding the dual objectives of CO_2 emissions and biodiversity– while enhancing CPO production and oil palm companies' income–can be achieved under a credit-based economy. More specifically, by firms making a more appropriate and sustainable use of credit facilities. Moreover, further international financial help is needed to compensate the lack of funding allocated by the Indonesian Government for conservation purposes – since governments from developing countries are usually more focused on reducing poverty and other social issues than on enhancing environmental unsustainability (Redfield, 1996). Therefore, a more equilibrated power distribution between conservation and financial forces– rather than creating contexts where either of them "take control" over the SES dynamics – is needed. Yet, whilst the model shows evidence that achieving a win-win-win context regarding CPO production, CO_2 emissions and biodiversity in Indonesia is potentially achievable in the short and medium-terms (2017-2050), it is considered necessary to perform sustainability analyses that go beyond this time-frame. In particular, to explore whether credit-driven SES, which are highly dependent on self-generating debt mechanisms, can be truly sustainable in the long-term. The following section is used to explore this issue and as a starting discussion point regarding the internal and external mechanisms for long-term (un)sustainability in Indonesia.

4.4.2 What factors enhance system rigidity and long-term (un)sustainability under debt-based economic systems?

The SF scenario revealed that it is possible to enhance synergies between the indicators selected by implementing specific policies when private financial (banks) and public governmental powers lie in equilibrium (see SF scenario, Figures 4.3, 4.4 and 4.5). It is asked, therefore, whether these results could also serve to enhance long-term SES sustainability in Indonesia under the context considered.

Sustainable development is potentially an important shift in understanding relationships of humanity with nature. This concept has become so comprehensive and complex that it is no longer useful in guiding policymaking (Holden *et al*., 2014). However, it can be a useful framework to explore de-coupling processes between economic and environmental elements in complex SES. Applied to the context of this chapter, long-term sustainability can be defined as a system's ability to persist over time (Dawson *et al*., 2010); in other words, sustainability occurs over an infinite time horizon in which the objective is to persist and maintain system functions, i.e. the goal is to continue to play the game (Carse, 1987).Although the aim of this chapter is not to use the modelling results beyond the selected time-frame (2017-2050), the previously performed analysis provides some insights regarding those characteristics that could hinder long-term sustainability in the SES modelled (Ulanowicz *et al*., 2009). In particular, the lack of capacity of the SES to cultivate internal autocatalysis and its high dependency on unstable external financial institutions. It is argued that these characteristics make the SES modelled a 'rigid' system (see Burkhard *et al*., 2011). System rigidity refers to a situation where a system becomes so efficient in its processes that there is little room for further innovation and sustainability (Fath *et al*., 2015). Characteristics of a rigid system include: very few key nodes and a high concentration of influence, being highly vulnerable to external disturbances because of reduced diversity (Fath *et al*., 2015) and brittleness, i.e. lack of resilience (Jackson, 2010).As shown by the model, Indonesia possesses a high dependency on unstable external financial institutions, with two main nodes (i.e. palm oil industry and banks) and a primary single pathway connecting both agents, "navigated" by credits and interests.

In particular, four main socio-economic and political factors could be currently strengthening and reinforcing system rigidity in Indonesia. First, Indonesia is the first exporter of palm oil in the world; only between 2000 and 2014, exports and consumption of CPO in Indonesia increased from 5 to 22Mt and from 3 to 11 Mt, respectively (USDA, 2014). The significant contribution of oil palm production to regional, national and local economies(Zen *et al*., 2005) will be supported by doubling the land area under oil palm by 2020 (UNDP, 2015). Second, the production of CPO has resulted in economic improvement of rural areas by providing jobs for local people (Hirawan, 2011). More specifically, increasing agricultural incomes from CPO production is critical to escape poverty for poor smallholder households that depend largely on natural resources for their livelihood (Klasen *et al*., 2013).Third, system rigidity is also enhanced by the reliance of the palm oil industry on upfront capital funding from overseas banks, needed to develop land, plant seedlings and build infrastructure (Chain Reaction Research, 2017). Thus, the current credit-based palm oil industry is supported by both banks and the industry itself, since it enhances a win-win economic context where the prior gain benefits from the interest on their loans and the latter continues to increase its turnovers due to the rising demand over CPO production. Moreover, the aversion to risk of banks and farmers, as well as the high operational costs, leaves little room for change in terms of carrying out more sustainable practices (Ruysschaert *et al*., 2011). Last, but not least, weak conservation governance in most tropical countries does not help to counterbalance system rigidity supported by the credit-driven palm oil industry. This places BAU economic forces at a privileged position at the expense of conservation forces (Hill *et al*., 2015a). As a result, developing countries, such as Indonesia, do not possess enough funding for conservation (or are not willing to use it for that purpose), nor receives enough international financial support. For instance, although Indonesia signed the US$1 billion deal with Norway under the REDD framework (Lang, 2010a), the agreement has not made much difference to the rate of deforestation so far – due to different reasons related to corruption, bad practices, and the stronger economic forces present in Indonesia compared to conservation (Lang, 2010a; Lang, 2017). See section 6.2.2 (Chapter 6) for a more detailed analysis on this case and PES schemes.

These factors create a context in Indonesia where the debt-driven and credit-dependent CPO production system will likely continue to be socio-economically supported in the long-term by the different actors and entities involved in the CPO production process, including banks, oil palm companies, farmers and the government. It is necessary to enhance a shift in the mainstream BAU thinking among palm oil stakeholders and farmers through novel farmer policy guidance, environmental legislation and incentive mechanisms. In this regard, besides the previously discussed PES schemes, favouring partial public (governmental) intervention in the CPO market system could help addressing the previously described factors and reduce system rigidity. For instance, market intervention through different policies could address the Indonesian smallholders' aversion to risk, currently represented by their unwillingness to use credit facility from overseas banks to create new plantations in degraded lands; hence, cheaper bank financing mechanisms (e.g. interest-free loans) offered by more secure financial entities, e.g. micro-finance institutions (see Ruysschaert *et al*., 2011) could incentivize a more sustainable use of bank credits by farmers. Similarly, stronger conservation governance could help compensate the negative environmental impacts exerted by the stronger financial powers driving land clearing in Indonesia. In fact, good conservation governance has proofed successful in reducing deforestation and the number of unprotected forests in some tropical areas, such as the Amazon (Soares-Filho *et al*., 2006). However, high corruption and low public governance quality in Indonesia, which was ranked second to last in a Global Competitiveness Report survey in 2015 (OECD, 2016), could be hindering long-term sustainability and system rigidity through low effective funding allocation for forest, wildlife and natural resource conservation (Sodhi *et al*., 2007). The problem here is the political difficulty of implementing policies that, indirectly, reduce the power of influential financial institutions that are not interested in any paradigm shift. Thus, governments are usually not free to create new institutions that could help enhance long-term sustainability at will, but must take account of the influence of industries and other interest groups (Abel *et al*., 2006). This could be due to the high dependency of national economies on very few corporations or monopolies In fact, this could be one of the reasons why systems so often remain maladapted to current unsustainable conditions, to the point of collapse (Abel *et al*., 2006). Developing countries within tropical regions would be, therefore, benefited from better conservation governance, as well as higher levels of public expenditure through international PES schemes and welfare programmes (Hopkin and Rodriguez-Pose, 2007). Thus, governments form developed countries need to assist these countries in the effort to achieve the sustainable natural resources under credit-based economic systems (Balmford *et al*., 2002).

4.4.3 Further research and areas for improvement

The model presented in this chapter has short and medium-term governance and policy implications to enhance sustainability in debt-based SES, using Indonesia as a case-study. Taking this research forward would require a more complex banking system, as well as more detailed credit lending mechanisms, in order to improve various model simplifications and assumptions. More specifically, to explore the extent to which overseas banks are willing to lend credits to firms to finance innovative CPO production processes (e.g. for technology efficiency improvements and degraded land upgrading), instead of traditional palm oil cultivation processes – since the latter is, in principle, a more secure financial investment for banks. Thus, there is a need to explore alternative banking mechanisms that enhance profits for both firms and banks while supporting environmental conservation. Moreover, conservation forces can be further specified by integrating empirical data from current specific PES schemes, which would be the drivers of environmental conservation in the SES modelled. Furthermore, carrying out participatory processes with stakeholders, including farmers and government agents, banks and oil palm companies involved in the CPO production process, would help building a more realistic SES. Collecting further data from the bottom-up would enable a more detailed modelling analysis regarding the relationships, adaptive behaviour and learning dynamics between agents and the environment modelled(see Bousquet and Le Page, 2004).

4.5 Conclusion

Modelling results (SF scenario) showed enhanced SES sustainability values under certain socio-economic and governance contexts. The alternatives – whereby economic growth has priority through oil palm expansion (BAU scenario) or the trading-off of biodiversity and CO_2 emissions are indirectly enhanced together with a partial decrease in CPO production (RBE and RCE scenarios) –implied substantial losses of biodiversity and CPO stocks, with increasing CO_2 emissions.

The conclusions from this chapter are threefold:

- *Economic-development forces are stronger than conservation forces in Indonesia* (BAU scenario). This situation is currently strengthened and reinforced by weak conservation governance. Similarly, by various socio-economic and political factors – related to the high dependency of the national, regional and local economies on (unsustainable) CPO production. This context enhances an inappropriate and unsustainable use of credit facilities by palm oil companies for funding CPO production.

- *SES sustainability can be enhanced, not by incurring a change or replacement of the current debt-based economic system or CPO production system in Indonesia, but by shifting the mainstream BAU thinking among key economic actors and entities – including palm oil companies, stakeholders and farmers*. Thus, shifting market-driven, capitalist forces to support environmental conservation – and balancing conservation and economic forces – requires novel farmer policy guidance, environmental legislation and incentive mechanisms from international bodies and developed countries. In this regard, the SF scenario showed the positive impacts of increasing technology efficiency in existing cultivations and creating new plantations in degraded land.

- Besides international financial help, *there is a need to enhance conservation governance in Indonesia*. Not only in terms of increasing protected areas and restoring degraded land, but also favouring partial, responsible governmental intervention in the CPO market system.

The question so far has been whether the Indonesian and global societies are prepared to either pay the financial and societal costs of withholding oil palm development, or accepting a comparatively smaller trade-off with agricultural land in return for increasing environmental sustainability. Yet, the analysis performed in this chapter shows that it is possible to pursue a course of sustainable development that substantially minimizes trade-offs in the short to medium-term. Whilst there is a need for models that analyse the long-term impacts of debt-driven SES, this model suggests a lesson for developing countries facing problems of poverty and unsustainable use of natural resources: environmental unsustainability can be defeated, at least temporally, by partially shifting the banking-capitalist forces to support environmental conservation, without abandoning the role of international bodies and the state in protecting the environment from the rough edges of the market economy.

Chapter 5:

Sustainable futures in tropical landscapes: A case-study in the Wet Tropics

> *"The cheapest and most efficient way of slowing down global warming is to protect and restore the forests, particularly the tropical forests*"
>
> – Jane Goodall (British primatologist, 2011, Jane Goodall Institute, Melbourne)

5.1 Introduction

Humans now manage the majority of land on earth, with more and more land allocated to agriculture, especially in tropical forests, which are declining (Venter *et al*., 2016). It is, therefore, no surprise that a debate about how to reconcile the needs of people and nature has resurfaced (Fenning, 2014). This question is particularly important in tropical regions, which face three main issues for sustainability. First, future food demand is projected to increase by at least 70% by 2050 in response to growing levels of per capita consumption, shifts to animal-based diets, and increasing population (Nelleman, 2009). Improving agricultural productivity in the tropics is critical to meet this demand (Fedoroff, 2010), as well as to alleviate chronic food insecurity currently affecting nearly one billion undernourished people (Nelleman, 2009). Second is the need to reduce atmospheric concentrations of GHG to address climate change that is progressively affecting agriculture, coastal areas, human health, and many other sectors (UNFCCC, 2009). International policy discussions have been focusing on reducing emissions from tropical deforestation and degradation (e.g. UN-REDD Programme) for climate mitigation (Angelsen, 2008). Third is biodiversity loss. The global biodiversity crisis has been well documented, with one-fifth of the world's assessed vertebrates being at imminent risk of extinction (Hoffman et a., 2014) and many more less studied species thought to be under similar threat (Tedesco *et al*., 2014).In tropical landscapes, LUC driven by the expansion and intensification of agriculture and plantations (Foley, 2005) is a main cause of biodiversity loss and ES (Harrison *et al*., 2014), resulting in areas affected by humans being less genetically diverse than wilder regions (Goulart *et al*., 2016).

How can we achieve the greatest conservation and climate change mitigation outcomes in a landscape given production demands for food, fibre, fuel, and other ecosystem services (ES)? This trade-off is generally addressed by two broad strategies at the landscape level: one intensifies farming to allow the offset of areas in which nature is protected – land-sparing (LSP) – while the other integrates agricultural production and nature protection in an agro-ecological matrix –land-sharing (LSH) (Green *et al*., 2005; Hulme *et al*., 2013; Phalan *et al*., 2011). The LSP versus LSH framework can be used to determine what balance of land-use intensity and conservation is needed in order to benefit both biodiversity (Gordon *et al*., 2016) and production outcomes, while considering carbon emission mitigation strategies.

This chapter presents an integrated Agent-based Model (ABM), built using NetLogo (Wilensky, 1999), which combines Geographic Information Systems (GIS), Bayesian Belief Networks (BBN), empirical data and expert knowledge in order to examine the impact of development, protection and restoration forces on the SES sustainability of the Wet Tropics Natural Resource Management (NRM) Region, Queensland, Australia. Model outcomes show the empirical and spatially explicit impacts of these LUC processes on the SES sustainability of the case-study – which, as explained in Chapter 1, refers to the capacity of a SES to enhance win-win-win outcomes regarding biodiversity conservation, climate change mitigation, and food production. Thus, the model studies the extent to which one provisioning ES (sugarcane production), one regulating ES (carbon sequestration) and biodiversity are affected. More specifically, model outcomes are used to address two main questions; namely [a]which land-use and landscape governance scenario (Business as Usual (BAU), LSP, or LSH) would increase SES sustainability in the Wet Tropics NRM Region; and [b] which governance contexts and combination of socio-economic factors could help limit the expansion of agricultural intensification while improving sustainability in those tropical SES driven by market economic forces.

5.2 Methodology

A full Overview, Design concepts, and Details (ODD) protocol, describing the model in detail is available in Appendix B (pp. 53-76).

5.2.1 Research objective and case-study area

An empirical and spatially explicit ABM is constructed to explore the effect of three future LUC scenarios (BAU, LSP, and LSH) on trade-offs and synergies among two different ES (carbon sequestration, sugarcane production) and biodiversity, in the Wet Tropics NRM Region for the period 2016-2030.The Wet Tropics NRM Region of northeast Queensland (Figure 5.1) covers an area of 21,722km^2 and is the only region to include two contrasting World Heritage Areas side by side – the Wet Tropics World Heritage Area (WTWHA) and the Great Barrier Reef (GBR). The area extends from Bloomfield in the north, south to Ingham and west to Mount Garnet, and includes the Atherton Tablelands. The area is home to both a rich and enduring Aboriginal cultural heritage and one of the most biologically diverse areas in the world, with forests embracing thirty-five international global biodiversity hotspots. Fifty per cent of current land in the Wet Tropics NRM Region is protected, a considerably larger area than the main industry – sugarcane production – which covers around eight per cent of the total region (DSITI, 2016). The current BAU context is, in effect, a “partial” LSP process, where protected areas have been increasing by around 20 per cent since 1999 with the area covered by sugar plantations remaining relatively stable (DSITI, 2016).

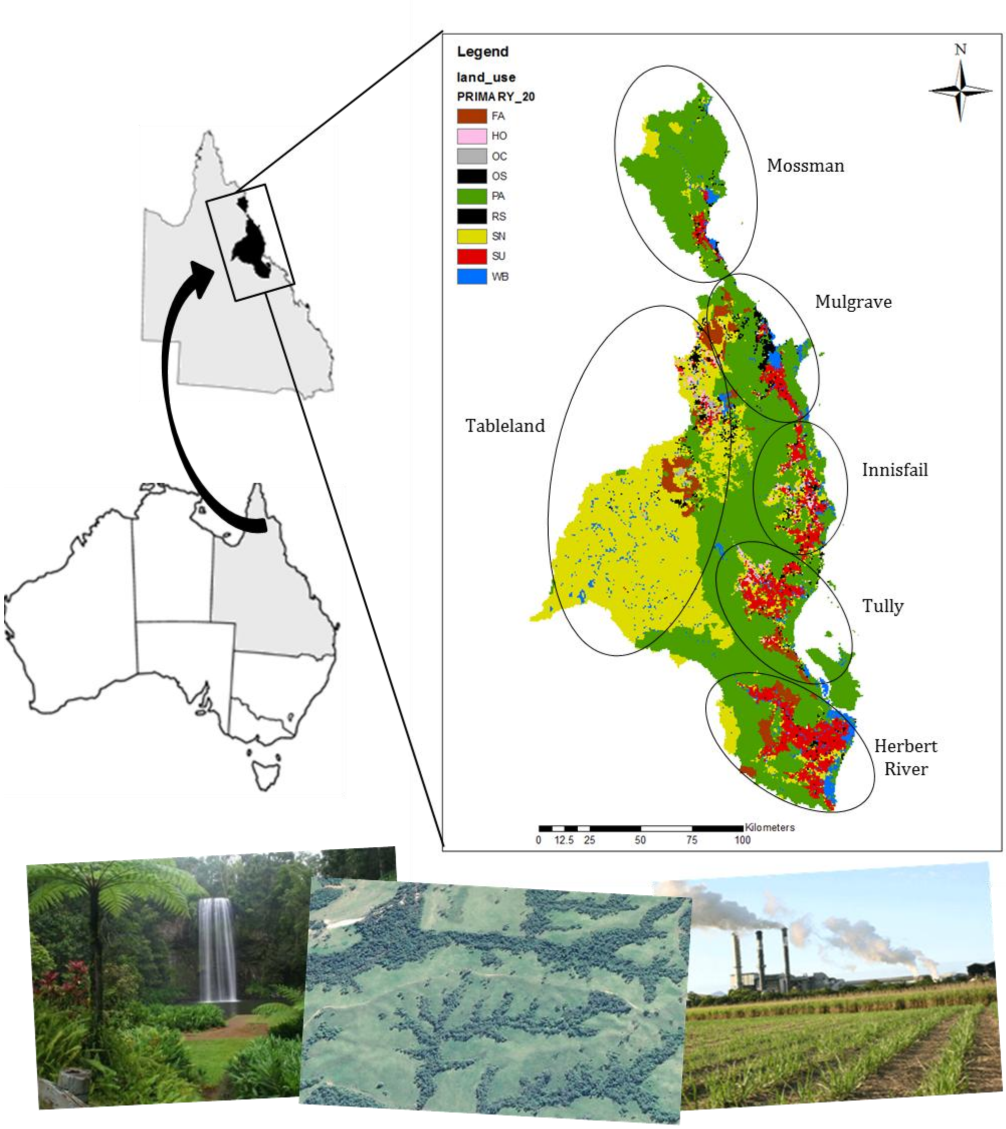

Figure 5.1: Geographic location of the Wet Tropics NRM Region, North East Queensland, Australia. The primary land-uses displayed by the map are: forestry areas (FA), horticulture (HO), other crops (OC), other services (OS), protected areas (PA), residential areas (RS), semi-natural areas (SN), sugarcane lands (SU) and water bodies (WB). Circled areas show the different sugarcane mill-areas present in the case-study area. The photographs on the bottom show local examples of the three primary land-uses considered for the analyses; namely protected areas (left), semi-natural areas (centre), and sugarcane lands (right).*Source*: author.

The model constructed is used to perform a trade-off analysis, where different scenarios are compared by analysing how specific indicators (equivalent for all scenarios) change over time based on different forces driving LUC. More specifically, the model explores the extent to which LUC forces related to conservation, restoration and production affect biodiversity, carbon sequestration and sugarcane production in this region. Although LSP versus LSH studies are usually focused on minimizing trade-offs between biodiversity and a production goal (sugarcane in this case), this research also includes carbon sequestration as an environmental indicator based on the importance, from an environmental perspective, of carbon emissions due to tropical deforestation (explained in Chapter 1). Sugarcane production is selected as a representative indicator of economic-development forces due to the sugarcane industry being one of the most important rural industries in Australia (AgriFutures, 2017), currently threatening the rich biodiversity of the North-East of Queensland through sugarcane plantation expansion. Hence, sugarcane production, carbon sequestration and biodiversity are the three main indicators examined under the LSP versus LSH framework in this model.

5.2.2 Spatially-explicit modelling of the land sharing/land sparing framework

The rationale behind selecting the LSP versus LSH approach, and the spatially-explicit nature of the model, are both interrelated. In particular, the particular geographical context of the case-study area (i.e. land-use distribution) is suitable for a spatially-explicit approach within the LSP versus LSH framework.

First, the current land-use distribution in the Wet Tropics – with almost 50 percent of land protected and a steady stable 8 percent of agricultural land allocated for sugarcane production (DSITI, 2016) – is considered reasonably 'locked-up', where LUC processes rarely occur and are less frequent than in most other tropical areas (see DSITI (2016)). As a result, the 2015 land-use distribution map for the Wet Tropics NRM Region (Figure 5.2) displays a clear spatial separation between semi-natural areas (left), protected areas (centre) and sugarcane lands (right). Besides the increase in the number and extent of protected areas, such clear (from a spatial perspective) land-use distribution has remained relatively stable over the last two decades (DSITI, 2016). This spatial separation between land-uses has also been enhanced by the characteristic environmental conditions of the Wet Tropics, where certain environmental factors – such as climatic conditions (e.g. rainfall) or soil potential to grow sugarcane – vary considerably from some areas to others within the region. This reinforces extreme environmental gradients along the landscape, thus enhancing specific distributions of land-uses (see section 5.2.5 'Data' below). Due to this, it is necessary that models exploring SES sustainability issues in the Wet Tropics produce spatially-explicit outputs, where the specific spatial location of any future land conversion and LUC processes taking place in this region becomes particularly relevant.

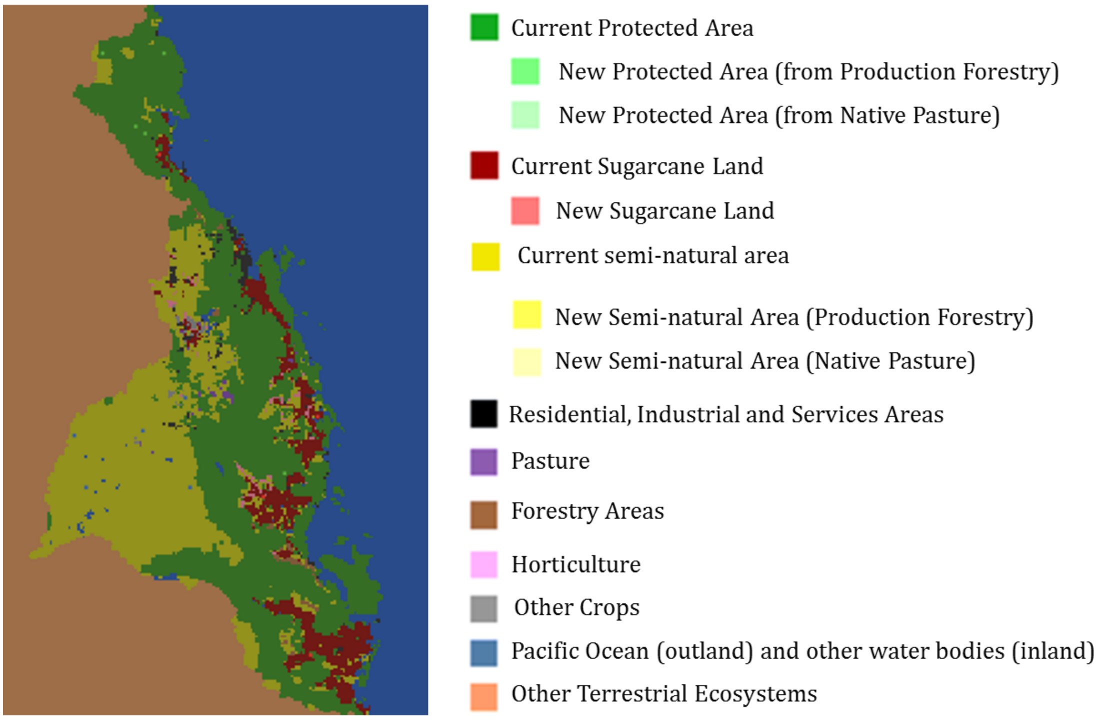

Figure 5.2: Initial (2015) primary land-use distribution for the Wet Tropics NRM Region, obtained by integrating a primary land-use and cover map (DAF, 2015) into NetLogo. Note that the case-study area is located between the Pacific Ocean (to the right, in blue) and other terrestrial ecosystems (to the left, in light orange), which are not considered for this research. The legend from Figure 5.2 is also used for Figure 5.5 below. *Source*: author.

As previously mentioned, this spatially-explicit context provides a suitable scenario to implement the LSP versus LSH framework in the Wet Tropics NRM Region. If one looks at Figure 5.2, the spatial separation between three primary classes of land-use provides a potential platform to apply the LSP versus LSH framework from a spatial perspective. In particular, there is an obvious spatial segregation between semi-natural areas (in yellow) and intensive sugarcane agriculture (in red), together with protected areas (in green). It is argued that the environmental and land-use characteristics of semi-natural areas in the Wet Tropics NRM Region align with the concept of LSH, while both sugarcane plantations and protected areas combined align with LSP. More specifically, the land-use classification developed by ACLUMP[12] (2016) refers to semi-natural areas as a primary class based on "production from relatively natural environments", defined as "land that is used mainly for primary production with limited change to the native vegetation". Thus, semi-natural areas include native forests and grasslands that are subject to relatively low levels of intervention, where the structure of the native vegetation generally remains intact despite deliberate use (ACLUMP, 2016). Considering that LSH, also called 'wildlife-friendly farming', is known as a land-use system that combines low intensive agricultural production with protection in an agro-ecological matrix (Green *et al.*, 2005; Hulme *et al.*, 2013; Phalan *et al.*, 2011), it is argued that semi-natural areas in the case-study area – as defined in ACLUMP (2016) – align with the concept of LSH. Similarly, protected areas and sugarcane land in Figure 5.2 are defined by the ACLUMP (2016) document as primary classes consisting of "conservation and natural environments" (including strict nature reserves, national parks, and other conserved areas) and "intensive sugarcane production from irrigated and dryland agriculture", respectively. Thus, the combination of both protected areas and sugarcane agricultural land is considered to align with the concept of LSP, which is based on intensifying production to maximize agricultural yield within a fixed area, while dedicating other land to biodiversity conservation (Green *et al.*, 2005; Hulme *et al.*, 2013; Phalan *et al.*, 2011).

In short, the empirical and spatially-explicit model constructed explores LUC dynamics with regards to semi-natural areas (LSH) and the nexus protected areas–sugarcane land (LSP).Furthermore, the spatially-explicit nature of the model aims to contribute to the lack of spatially-explicit LSH vs. LSP studies (Fischer *et al.*, 2014; Law *et al.*, 2015). In particular, the model is used to explore the extent to which different future LUC scenarios affect SES sustainability, i.e. biodiversity, carbon sequestration and sugarcane production, under the LSP versus LSH framework. The LUC scenarios explored, which correspond to the period 2016-2030, include LSH (where semi-natural areas increase/decrease), LSP (where protected and sugarcane areas increase/decrease) and

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[12] ACLUMP stands for Australian Collaborative Land Use and Management Program Partners. This nationally consistent document provides a land use nomenclature and classification scheme for Australia, which involves ordering land use in a systematic and logical way.

BAU (where protected areas increase at the same rate as during the period 1999-2015 (see DSITI, 2016).Table 5.1 below shows a qualitative description of the rationale for the different scenarios modelled.

Table 5.1: Narratives of the scenarios modelled for the period 2016-2030. *Source*: author

Scenario (2016-2030)	**Description**
Business As Usual (BAU): "World Heritage"	The number and extent of protected areas in the Wet Tropics NRM Region keep increasing, in order to meet conservation targets as a World heritage listing site. The total extent of semi-natural areas increases slightly following the trends from the period 1999-2015. Production (mainly sugarcane) remains stable over time due to other regions in Queensland (e.g. Mackay-Whitsundays) being more focused on meeting national production demands.
Land Sparing (LSP): "World Heritage and Queensland's 'food bowl' region"	The region continues to meet conservation targets by increasing the number and extent of protected areas. However, this is combined with increases in the amount of land focused on agricultural (sugarcane) production, enhanced by the Queensland and Australian governments. The goal is the Wet Tropic NRM Region to improve its contribution to the so-known Australian 'food bowl' process.
Land Sharing (LSH): "Multifunctional landscapes"	Queensland and Australian Governments lead a transition towards more multifunctional discourses and governance framework, where wildlife-friendly farming practices are enhanced at the expense of lower sugarcane yields. Thus, the Wet Tropics NRM Region follows opposite trends than in the LSP scenario, where both protected areas and sugarcane lands decrease in exchange of semi-natural areas (above all production forestry).

5.2.2 Modelling framework

The ABM presented in this chapter can be considered an Agent-Based Land-Use Model (ABLUM) (see Matthews *et al*., 2007; Polhill *et al*., 2011), which combines Bayesian Belief Networks (BBN), Geographic Information Systems (GIS), empirical data and expert knowledge. NetLogo (Wilensky, 1999) is used as the ABM construction software. Besides the results obtained from modelling scenarios, the integrated modelling approach selected aims to contribute to one main demand within the ABM community (O'Sullivan *et al*., 2016), based on building hybrid ABMs, which are based on the integration[13] of different modelling techniques to reconcile the advantages of different approaches. Here, BBNs and GIS layers are integrated into a NetLogo ABM through reporters including if-else procedures (see 'Bayesian Belief Networks' section for more information).

5.2.3 Entities, state variables and scales

The key entities in the model are agents, which represent power governance forces(i.e. *PG-agents*) driving LUC (i.e. conservation, restoration and sugarcane production); and patches, which represent land-uses (A). *PG-agents* are classified in three types, following the previously described LSP versus LSH framework: PG_d (forces driving development of land for sugarcane production, i.e. LSP), PG_p (governance forces driving the creation of new protected areas, i.e. LSP), and PG_{mr} (governance forces driving restoration and maintenance of semi-natural areas, i.e. LSH). Land-uses are classified in three types: A_p (protected areas), A_a (semi-natural areas), and A_d (sugarcane areas). Semi-natural areas are classified in A_{ag} (native pasture) and A_{ap} (production forestry). Figure B–3.1, Appendix B (p. 54), shows a Unified Modelling Language (UML) class diagram describing the model entities and variables in detail. TablesB–3.1,

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[13] Note that the 'integrated' terminology used in this chapter does not refer to the actual integration of an ABM software (i.e. NetLogo) into a GISystem software – nor the other way around. Rather, it refers to the use of spatial data (i.e. GIS layers) in NetLogo, in order to produce the spatial outputs shown in the Results section.

B–3.2 and B–3.1 in Appendix B (pp. 57-58) show a description of the entities and state variables modelled, their units and data sources.

The spatial-scale of the model is considered to be regional, due to the case-study area being a Natural Resource Management (NRM) region. Thus, the model works at a finer scale than other spatially explicit models and studies focused on ES at the global (Haines-Young, *et al*., 2012; Maes *et al*., 2012; Naidoo *et al*., 2008; Turner *et al*., 2007) and national (Egoh *et al*., 2009) levels. In this regard, regional scales are considered more operational in policy-making compared to larger or smaller spatial levels (Wi, 2013). Thus, the regional scale selected is directly relevant to the management of the Wet Tropics NRM Region, due to this area being managed at the level of the World Heritage Area through the Wet Tropics Management Authority. At the same time, *PG-agents* (i.e. forces driving LSH and LSP through land protection, restoration and clearing processes) are also modelled at the regional level. In particular, PG_d*-agents*, yet not modelled as actual farmers owning land parcels, represent agricultural expansion from smallholders, which is the main land clearing process occurring at the landscape/regional level in the Wet Tropics (Hill et al., 2015). Thus, PG_d*-agents* represent all the different forces driving land clearing for sugarcane production at the regional scale. PG_p*-agents* and PG_{mr}*-agents* are also modelled at the regional level, thus representing protection and restoration strategies implemented at the World Heritage Area level (i.e. regional scale) by the Wet Tropics Management Authority (Hill et al., 2015).

The time-scale of the model was selected based on expert knowledge. To decide how many time steps corresponds to one year in the model, experts used historic LUC data from the Department of Science, Information Technology and Innovation (DSITI, 2016) of Queensland. First, the yearly average change (in percent values and hectares) regarding the three main LUC modelled (i.e. protection, restoration and land clearing) was calculated for the period 1999-2015. Second, preliminary model outputs from the BAU scenario were analysed in order to know how many model time steps were needed to simulate the above-noted yearly LUC values. As a result, it was decided that LUC processes occurring in 20 time steps in the mode correspond to one year in the real world; thus, after 300 time steps the model is considered to have simulated 15 years, with 2016 and 2030 as initial and final years, respectively.

5.2.4 Simulation process and overview

Figure 5.3 shows a UML activity diagram representing the main dynamics of the system, and the flow from one process to the next one. The following is a list of the model processes taking place every time step, which are described in detail below (model functions and algorithms are described in the 'Submodels' section, Appendix B(pp. 73-76)): (i) Scenario selection; (ii) 'other land-uses' compute LUC; (iii) Patches compute LUC-suitability (SV) values from BBNs; (iv) PG-agents compute movement based on SV-values; (v) PG-agents compute PR-value and patches compute LUC; (vi) Patches compute indicators.

The environment consists of a grid of land-uses, where *PG-agents* move around the landscape representing the selected forces driving LUC. Computation of LUC follows the next rationale: each patch computes a total of three LUC suitability values (*SV-values*) per time step, one for each type of LUC (protection, restoration, and development). Thus, each patch computes one value for SV_p (suitability of the patch, if unprotected, to become a protected area, or to remain as protected if already protected), another for SV_{mr} (suitability of the patch to be converted to semi-natural land, or to remain as semi-natural if already semi-natural), and SV_d (suitability of the patch to be converted to sugarcane, or to remain as sugarcane land if already a crop). Thus, *SV-values* state the probability of each patch (land-use) to be converted to another land-use, or to remain the same. *SV-values* are obtained from Geographical Information Systems (GIS) and Bayesian Belief Networks (BBN) (explained below), and vary from one scenario to another.

Every time step, each *PG-agent* selects the patch with the highest compatible (to this *PG-agent*) *SV-value*. Thus, PG_p*-agents* only search for SV_p*-values*; PG_{mr}*-agents* only for SV_{mr}*-values*; and PG_d*-agents* only for SV_d*-values*. Moreover, each type of *PG-agent* selects the patch with the largest number of neighbour patches corresponding to that *PG-agent* type (this is computed to enhance patch connectivity). Hence, PG_p*-agents* seek for patches with more A_p patches around, *PG_d-agents* for A_d, and PG_{mr}-agents for A_{mr}. If there are no patches of the same type in neighbouring patches, the searching 'radius' is increased until patches of the same type are found.

Based on these 'rules', every time step each *PG-agent* will select one single final patch, called *target-patch*. *PG-agents* (i.e. LUC drivers) then move to their corresponding target-patch and compute one random-float number between 0 and 1, called *PR-value*: if the value lies between 0 and the *SV*-value, LUC in this patch is computed. Hence, the higher the *SV-value* in one patch, the higher the probability of this patch to compute LUC. If the value does not lie between 0 and the *SV-value*, the patch keeps its current land-use. This cycle is computed every time step for each *PG-agent*, thus driving model outcomes. In particular, regardless of whether LUC takes place in one patch or not, each patch computes different sugarcane production, carbon sequestration and biodiversity algorithms every time step. Note that there are no associated costs to agents' movement, since *PG_d-agents*, *PG_p-agents* and *PG_{mr}-agents* represent conceptual (theoretical) forces driving LUC, and not specific schemes or policies (that could be specifically accounted).

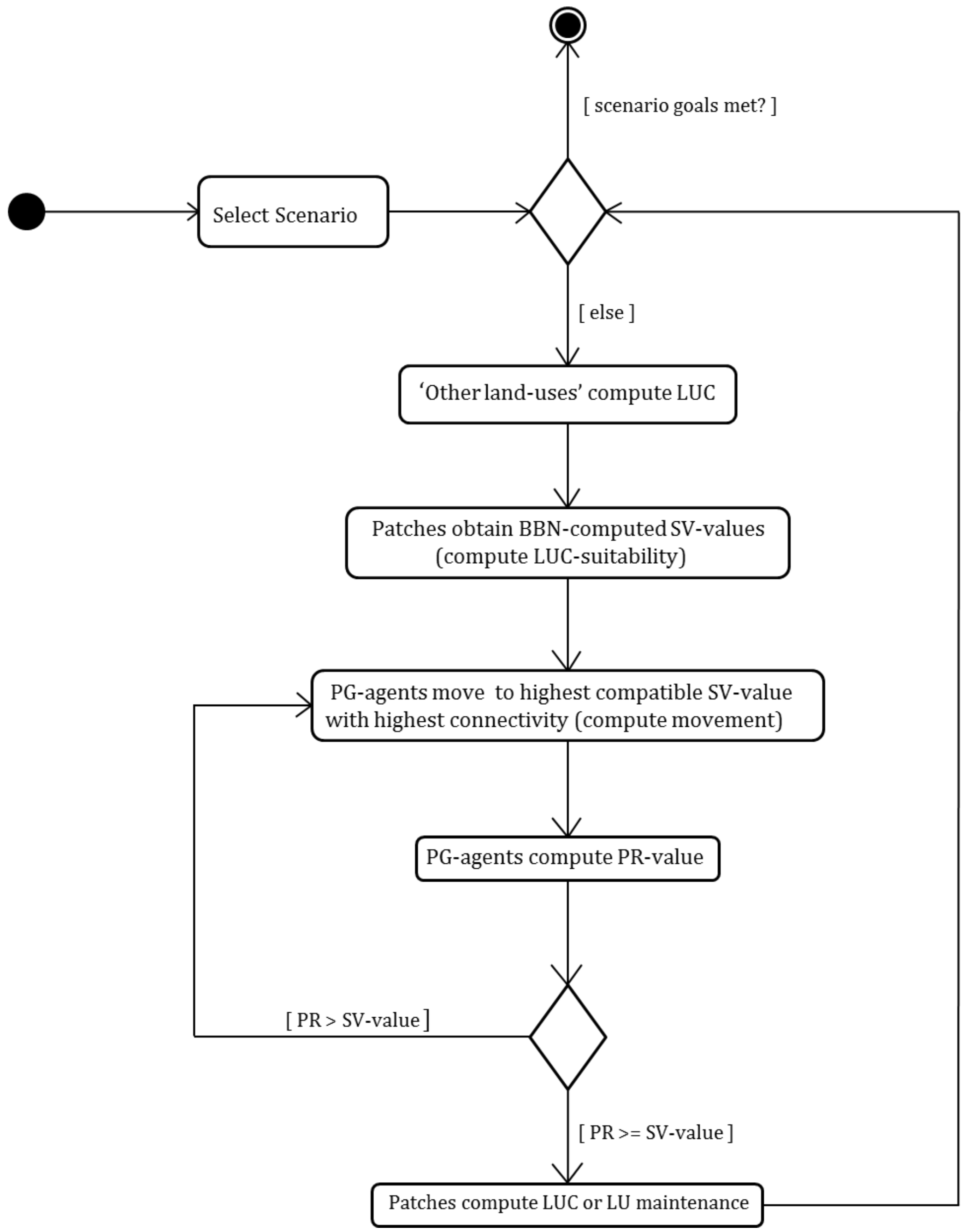

Figure 5.3: UML Activity Diagram. Structure diagram showing the step by step process computed by *PG-agents* and patches in the model. *Source*: author

5.2.5 Data

GIS layers and empirical data

The initial value of most of the parameters modelled (see Table B–3.4, Appendix B, pp. 60-63) are obtained from input GIS layers (see Table B–3.5, Appendix B, pp. 64-66). For those indicators with no GIS layers available, published empirical data was initially computed at the patch level, such as the initial sugarcane yield, sugar price, and carbon price (explained below). With regard to input data from GIS layers, because the spatial environment in NetLogo is in essence raster-based, NetLogo establishes a unitary data processing platform in order to import vector-based files (i.e. shapefiles) (Liu, 2015). Thus, when importing GIS Layers into NetLogo, the properties (attributes) of the vector files are copied to the properties of the NetLogo patches in order to be able to conduct data computation in the environment of NetLogo (see Appendix B, p.59, for a technical description about how to import GIS layers into NetLogo).

In this model, both ArcGIS and QGIS software were used to import GIS layers into NetLogo. First, the primary land-cover map for the Wet Tropics NRM Region (DAF, 2015) was used as a baseline and imported as a vector file into NetLogo. This process provided with an initial distribution of land-uses for the case-study area – i.e. the land-use map shown in Figure 5.1 was imported into NetLogo, thus obtaining the map shown in Figure 5.2 (the latter is a snapshot from NetLogo's Graphical User Interface (GUI)). As a result, each patch in the NetLogo model covered an area of 123.64ha of the case-study area. Out of the eleven total number of land-use types present in the Wet Tropics NRM Region (see Figure 5.2), only four are considered for the research analysis – yet these cover 97 per cent of the total land in the Wet Tropics NRM Region. These include rainforest (protected areas), native pasture & production forestry (semi-natural areas), and sugarcane land (developed areas). The focus on these land-uses follows the LSP versus LSH rationale explained at the beginning of the Methodology section, as well as the conceptual model developed by Hill *et al*. (2015a), which selects these land-use classes to perform a similar research study. The LUC for the remaining seven land-uses (hereafter called 'other land-uses', see UML diagram in Figure 5.3) are not analysed in the Results section, although their LUC processes are still computed for the sake of realism (see below).

Initial biodiversity values were obtained from a biodiversity GIS layer (Mokany *et al*., 2014) that aligns with the World Heritage criteria (UNESCO, 2016; WTMA, 2016).

Thus, each patch was given an initial biodiversity value, which varies over time following an 'habitat destruction-restoration (DR) function' (see p. 67, Appendix B).This function calculates the biodiversity for each patch based on the effect on the habitat quality of that patch, as well as the surrounding ones, of the above-noted LUC processes (see Tables B–3.6 and B–3.7, Appendix B (p. 68), for the value calculation process of the DR function). As for initial carbon sequestration values, both in tons and monetary value, these were obtained from an above-ground[14] biomass GIS layer (DE, 2004) – where the carbon conversion factor recommended by the Intergovernmental Panel for Climate Change for tropical Forests (IPCC, 2006) was used – and from the price corresponding to the 2013-2014 financial year (i.e. 24.15 AUD/t), respectively. Similarly to biodiversity, carbon values vary over time based on the impact of LUC processes taking place in each patch and the surrounding matrix of patches (see Table B–3.9, Appendix B (p. 69), for the function value calculation process). Initial sugarcane yield values were obtained as empirical (non-spatial) data from the Canegrowers Annual Report (Canegrowers, 2016), which shows historical data for the period 2006-2014. Regarding initial sugar monetary values, data for the period 2016-2020 (QSL, 2016) was extrapolated to the model simulation period (2016-2030), with an additional integrated random variability aimed at representing the highly volatile sugar price. See Table B–3.4, Appendix B (pp. 60-63), for the initial values and descriptions for the main variables computed in the model.

The *PG-agents'* movement is driven by both empirical data and GIS layers. As previously mentioned, *SV-values* are used by *PG-agents* to compute LUC in each patch. The data for this purpose is obtained from GIS raster files, where NetLogo converts the values from the attribute tables of these maps into patch variables. Table B–3.6, Appendix B (pp. 64-66), shows the different GIS layers used to set the *SV* values, including conservation potential (Mokany *et al*., 2014), sugarcane production potential (DAF, 2013a), rainfall potential (CSIRO, 2003) and carbon sequestration potential (DE, 2004). Maps regarding potential hardwood and softwood plantation (DAF, 2013b), potential intensive livestock and potential pasture (DAF, 2013c), potential perennial and

[14] Note that values for below-ground biomass were not included in the model due to lack of data in GIS format.

annual horticulture (DAF, 2013d) and priority urban development areas (DILGP, 2016) are used to obtain the LUC probability values regarding the 'other land-uses'. As previously mentioned, LUC for these other patches (e.g. rivers, pasture, cities) is modelled (without agents), yet the results are not analysed.

Bayesian Belief Networks (BBN)

BBNs represent the link between patches and *PG-agents*, thus providing essential information for *PG-agents*' LUC decision-making. The *SV-values* computed by each patch state the probability of that patch to change its land-use and cover, based on the above-noted GIS data, i.e. rainfall, conservation, sugarcane production and carbon sequestration potential. However, before *PG-agents* use *SV-values* to compute (or not) LUC, BBNs are constructed in order to set a relationship between the four different GIS layers. Thus, BBNs are used by *PG-agents* to answer questions such as the following: *is a patch (land-cover) with high conservation potential and low production potential suitable to be protected, under a BAU scenario?* or*: is a patch with moderate carbon sequestration potential and low conservation potential suitable to be converted into sugarcane plantations, under a LSP scenario?* BBNs, therefore, consider all the combination of variables from the GIS layers, and provide probability answers based on expert opinion. At the end of each time step, the BBNs compute one final LUC probability value (*SV-value*) per patch, which is thereby used by *PG-agents* to compute LUC:

A BBN is a graphical representation of a set of variables (nodes) and their causal relationships (links), forming a directed acyclic graph (Charniak, 1991). Nodes represent system variables, such as biodiversity or sugarcane yield, while links represent causal probabilistic relationships between two nodes. Within a BBN, each node has a defined set of states/categories along with a Conditional Probability Table (CPT) (see Figure 5.4for an example), which defines, for each category, the probability of it occurring given all possible category combinations from the(parent) nodes

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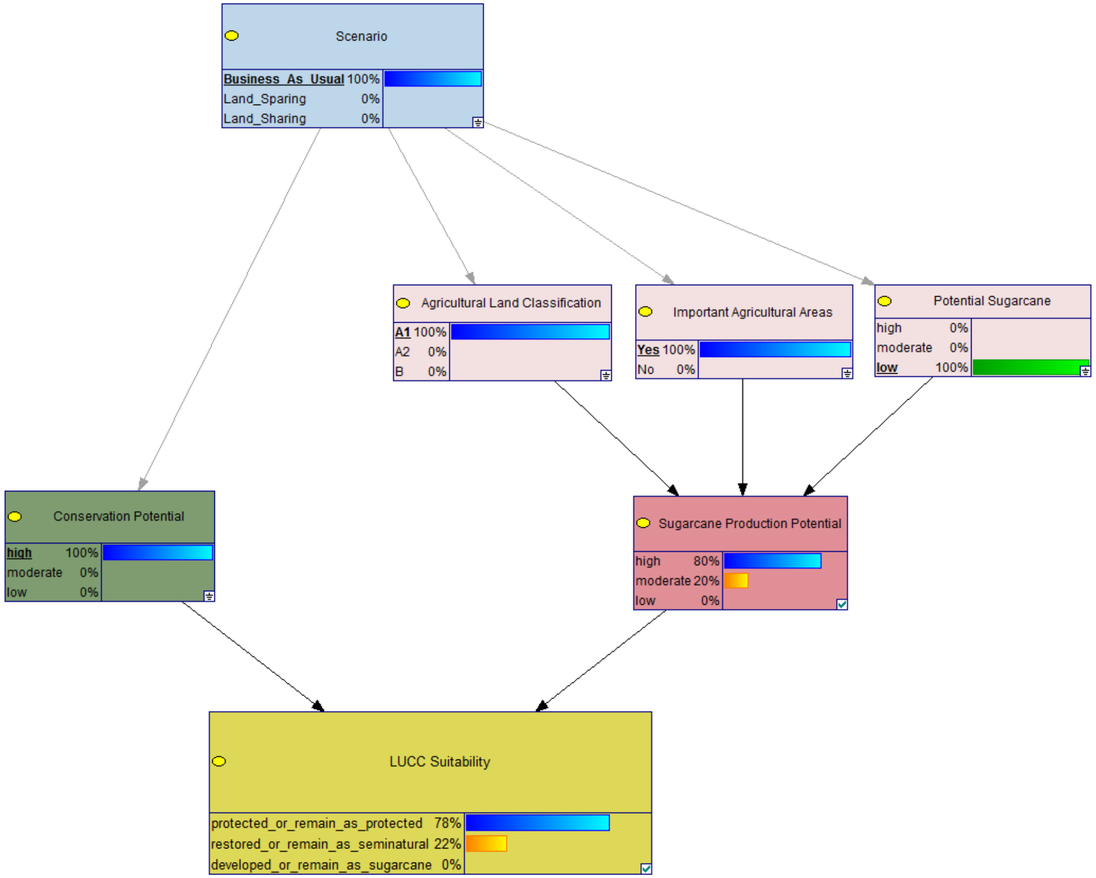

Conservation Potential	high			moderate			low		
Sugarcane Production Potential	high	moderate	low	high	moderate	low	high	moderate	low
protected_or_remain_as_protected	0.7	0.85	1	0.55	0.6	0.65	0.4	0.45	0.5
restored_or_remain_as_seminatural	0.3	0.15	0	0.45	0.4	0.35	0.6	0.55	0.5
developed_or_remain_as_sugarcane	0	0	0	0	0	0	0	0	0

Figure 5.4: Bayesian Belief Network (BBN). Example of a BBN developed using GeNIe®, with a Conditional Probability Table (CPT) on the bottom. Both dark red and green boxes represent biophysical spatially explicit (i.e. GIS) nodes, while light red and yellow nodes are completed using expert knowledge. Coloured bars represent the conditional probabilities for each CPT category. This particular BBN example is computed by semi-natural patches under the BAU scenario. In this particular case, the probability for one semi-natural patch to be protected, having 100 per cent of 'Conservation Potential' and 80 per cent of 'Sugarcane Production Potential', is 78 per cent, being the probability to remain as semi-natural 22 per cent, and to become developed 0 per cent. Due to 78 being higher than 22, the prior would computed as *SV-value* for this specific land-use. *Source*: author.

Although the use of BBNs for modelling LUC is not new (Bacon *et al*., 2002; Lynam *et al*., 2002), examples of the incorporation of BBNs to spatial ABMs are scarce and confined to utility maximization (Lei *et al*., 2005) or calibration of cellular automata transition rules (Kocabas *et al*., 2013).BBNs can help addressing uncertainties regarding agents' decision-making (Perez-Minana, 2016) within ES trade-off analyses (Gonzalez-Redin, *et al*., 2016). The BBNs constructed were built using the GeNIe (GeNIE and SMILE, 1998) BBN builder tool, while the BBN building process followed a logical framework adapted from the Australian Department of the Environment, Heritage and the Arts (DEWHA, 2010).

In the ABM, BBNs provide *PG-agents*' information to compute LUC. BBNs are integrated in NetLogo through reporters that compute tables including the BBN probabilities. These probabilities, which are completed using GIS layers and expert opinion, state the potential of each land-cover to be converted to another land-use, thus driving *PG-agents*' LUC decision-making. Likewise, BBN probabilities are updated every time step based on *PG-agents*' LUC decision-making (see section below). One BBN is created for each analysed land-use type (i.e. protected, semi-natural and sugarcane), where *PG-agents* of the same type compute the same BBN. Thus, nine total BBNs are computed (three BBNs per scenario), where every BBN has the same structure and nodes as the one in Figure 5.4. The probabilities for each CPT change every time step, whose initial values are set based on a product between expert opinion and GIS data. While the CPT categories from the input nodes (e.g. 'Agricultural Land Classification' in Figure 5.4)reproduce the attributes from the GIS layers – thus no expert-based interpretation is needed for their completion – the CPTs from intermediate (i.e. 'Sugarcane Production Potential') and output (i.e. 'LUC Suitability') nodes are completed using expert opinion. Finally, the 'LUC Suitability' output node has three different categories, one for each type of LUC (protection, restoration and development). With a value between 0-1, each category from this output node reveals the probability for each LUC type to take place in each patch every time step (called *SV-values*).

The following paragraph explains *PG-agents'* rationale for using the BBNs: Every time step, each *PG-agent* selects the patch with the highest compatible (to this *PG-agent*)

SV-value. Thus, *PG_p-agents* only search for *SV_p-values*; *PG_{mr}-agents* only for *SV_{mr}-values*; and *PG_d-agents* only for *SV_d-values*. Moreover, each type of *PG-agent* selects the patch with the largest number of neighbour patches corresponding to that *PG-agent* type (this is computed to enhance patch connectivity). Hence, *PG_p-agents* seek for patches with more A_p patches around, *PG_d-agents* for A_d, and PG_{mr}-agents for A_{mr}. As a result, every time step each *PG-agent* selects a final patch (i.e. *target-patch*), where they move and compute one random-float number between 0 and 1, called *PR-value*: if the value lies between 0 and the *SV*-value, LUC in this patch is computed (the *PR-value* is computed for *PG-agents* to be able to use the probabilities from the BBNs). Hence, the higher the *SV-value* in one patch, the higher the probability of this patch to compute LUC. This entire process is repeated each time step by each *PG-agent*. As a result of LUC taking place in each patch, biodiversity, carbon sequestration and sugarcane production values change, which are the model outputs analysed in the Results and Discussion sections.

Expert knowledge

The probabilities of those CPTs from intermediate and output BBN nodes (Figure 5.4) were completed using expert opinion. In order to gather expert-based qualitative data, the 'focus groups' method was used (Kitzinger, 1994; Morgan, 1998; Gill *et al*., 2008). Thus, groups of discussion were organized during one week in order to collect and integrate empirical information from different experts. Expertise was sought from five different scientists from a number of fields, including ecology, agricultural sciences, environmental governance, ecological economics and sustainability science. Thus, the objective was to establish conservation and production probability values for the 'Sugarcane Production Potential' and 'Conservation Potential' nodes, based on different scenarios and set of probabilities from other parent nodes. More specifically, during the first two days, experts provided an overall diagnostic of the situation in the case-study area through different discussions and by answering to open-ended questions. At a second stage, experts provided direct input to the above-noted nodes, where different combinations of probabilities were continuously integrated in the BBN in order to obtain different sets of preliminary results. These preliminary results were adapted and analysed, until the final set of probabilities for the intermediate and output BBN nodes were selected.

Furthermore, the target values for each scenario and the model's time scale (i.e. the number of years corresponding to each modelling time step) were also established through expert knowledge and discussed during the focus group meetings (see section 5.2.3 'Entities, state variables and scales' above).

5.2.6 Data discretization in GIS layers

Values from each GIS layer are clustered into different categories (e.g. high, moderate, low) using 'Jenks Natural Breaks Optimization' method in ArcGIS (Jenks, 1967). This method is particularly used under certain statistical conditions, such as relatively high variance between values (McMaster, 1997). This is the case for this model, where the parameter values selected for each scenario show a high variance. In particular, this value categorization method reduces the variance within classes and maximizes the variance between classes (McMaster, 1997). As a result, the values contained in each class are as similar to each other as possible while the mean of each class differs from the rest of classes to the highest extent.

5.2.7 Sensitivity analysis and Run setup summary

An OFAT (One-factor-at-a-time) sensitivity analysis was performed (ten Broeke *et al.*, 2016). Two varying parameters were selected, namely '*Initial* PG_p*-agents*' and '*Initial* PG_d*-agents*', while the values for the rest of parameters were selected based on literature review, empirical data, GIS information or expert opinion. Ten values (from 5 to 50) were selected for each '*Initial* PG_p*-agents*' and '*Initial* PG_d*-agents*' parameters, under each of the three scenarios (BAU, LSP and LSH), thus making300 possible combinations of parameters that, repeated 10 times each, made3000 runs overall. The sensitivity analysis is integrated within Figure 5.8in the Results section, where the impact of different magnitudes of economic and conservation forces (i.e. number of '*Initial* PG_p*-agents*' and '*Initial* PG_d*-agents*') on biodiversity is displayed.

5.3 Results

Results regarding the indicators selected are obtained for each of the three scenarios (BAU, LSP, LSH), and grouped in spatial and empirical impacts. A qualitative analysis was performed; this was due to the main interest in exploring the overall differences, in trends, among the indicators and scenarios tested – rather than the specific statistical significance of the results. Thus, similarly to the model presented in Chapter 4, a statistical analysis of the model would have not have contributed with any relevant information and data. The decision was supported by the high differences obtained among the indicators and scenarios modelled, thus discarding the need for a statistical analysis.

5.3.1 Estimated spatial impacts

Figure 5.5 shows the spatial explicit outputs obtained with NetLogo. Three output maps are obtained for each scenario, one for each time step (year) – 2020, 2025 and 2030 – thus obtaining nine maps in total. Note that the legend from Figure 5.2 – which showed the initial land-use distribution in the case-study area – is also used to describe Figure 5.5.

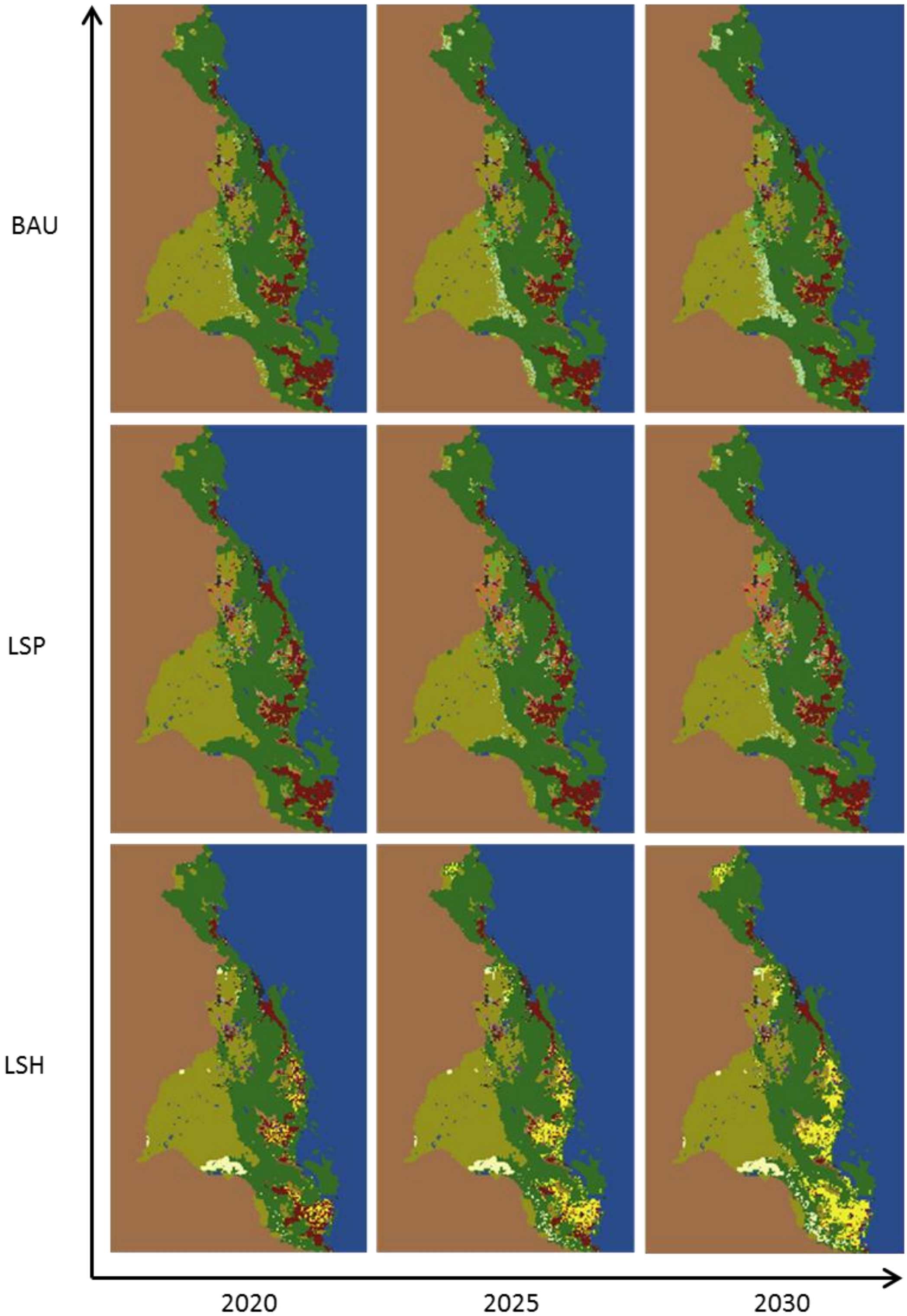

Figure 5.5: Spatial scenario outputs. Land-use variations are shown for each scenario (BAU = Business As Usual; LSP = Land-Sparing; LSH = Land-Sharing) regarding the years 2020, 2025 and 2030. Note that the legend from figure 5.2 has to be used for this figure. *Source*: author

Business As Usual (BAU)

The top row of Figure 5.5shows the spatial distribution of land-uses for the BAU scenario within the study area. In this scenario, protected areas increase in 10%in order to meet conservation targets as a World Heritage listing site, while production (mainly sugarcane) remains stable over time due to other regions in Queensland (e.g. Mackay-Whitsundays) being focused on meeting national production demands. The most spatial noteworthy trend is based on those semi-natural areas (i.e. native pasture and production forestry) with low sugarcane production potential and high conservation potential values being converted into protected areas – mainly located to the west of currently protected rainforests.

Land Sparing (LSP)

Queensland and Australian governments lead a transition towards a more multifunctional discourse (i.e. LSH), where wildlife-friendly farming practices (i.e. semi-natural areas) are enhanced (30%) at the expense of sugarcane yields and protected areas. Figure 5.5, in the middle row, shows the spatial distribution of new protected areas and new sugarcane lands converted from semi-natural areas. Semi-natural areas with high conservation potential and low sugarcane production potential values have a higher probability to be protected, while those with high production potential and low conservation values have a higher probability to be developed (for sugarcane production). New sugarcane areas are mainly located to the East of the Tablelands, with smaller areas in Innisfail, Tully and Herbert River.

Land Sharing (LSH)

The Wet Tropics NRM Region continues to meet conservation targets by increasing protected areas by 5%, yet combined with increases in sugarcane production by 22%. In Figure 5.5, the maps in the bottom row show new semi-natural areas converted from previously protected and sugarcane lands. While new native pasture areas are mainly converted from previously protected rainforests with low conservation value

(Tablelands), new production forestry areas are converted from both previously protected areas and sugarcane lands with low conservation and production potential values, respectively, located to the centre-east of the study area, i.e. Innisfail, Tully and Herbert River mill-areas (see Figure 5.1 for the distribution of the mill areas).

5.3.2 Estimated impacts

Figure 5.6 shows the empirical graphical results from the SES sustainability indicators selected, while Figure 5.7 shows the LUC trends for each scenario.

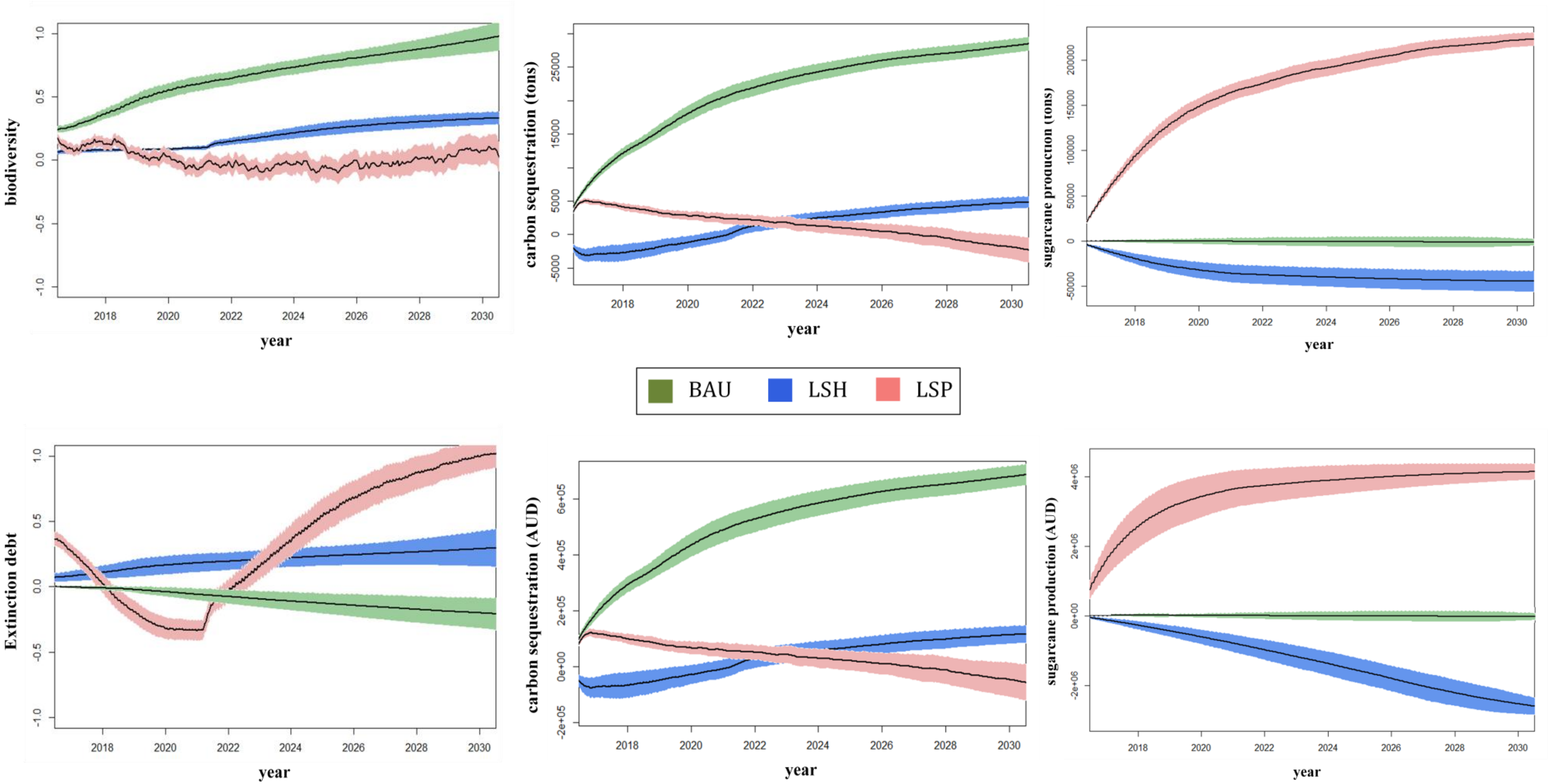

Figure 5.6: Graphical scenario outputs. Results are shown as the temporal variation (in net gains & losses) of different social-economic and environmental indicators for each scenario: BAU = green; LSP = red; LSH = blue (see legend). Both sugarcane production and carbon sequestration are shown in tons and Australian Dollars (AUD). Colour bands represent the standard error bands regarding all the runs computed for each indicator under every scenario. The black coloured lines show the mean values. *Source*: author.

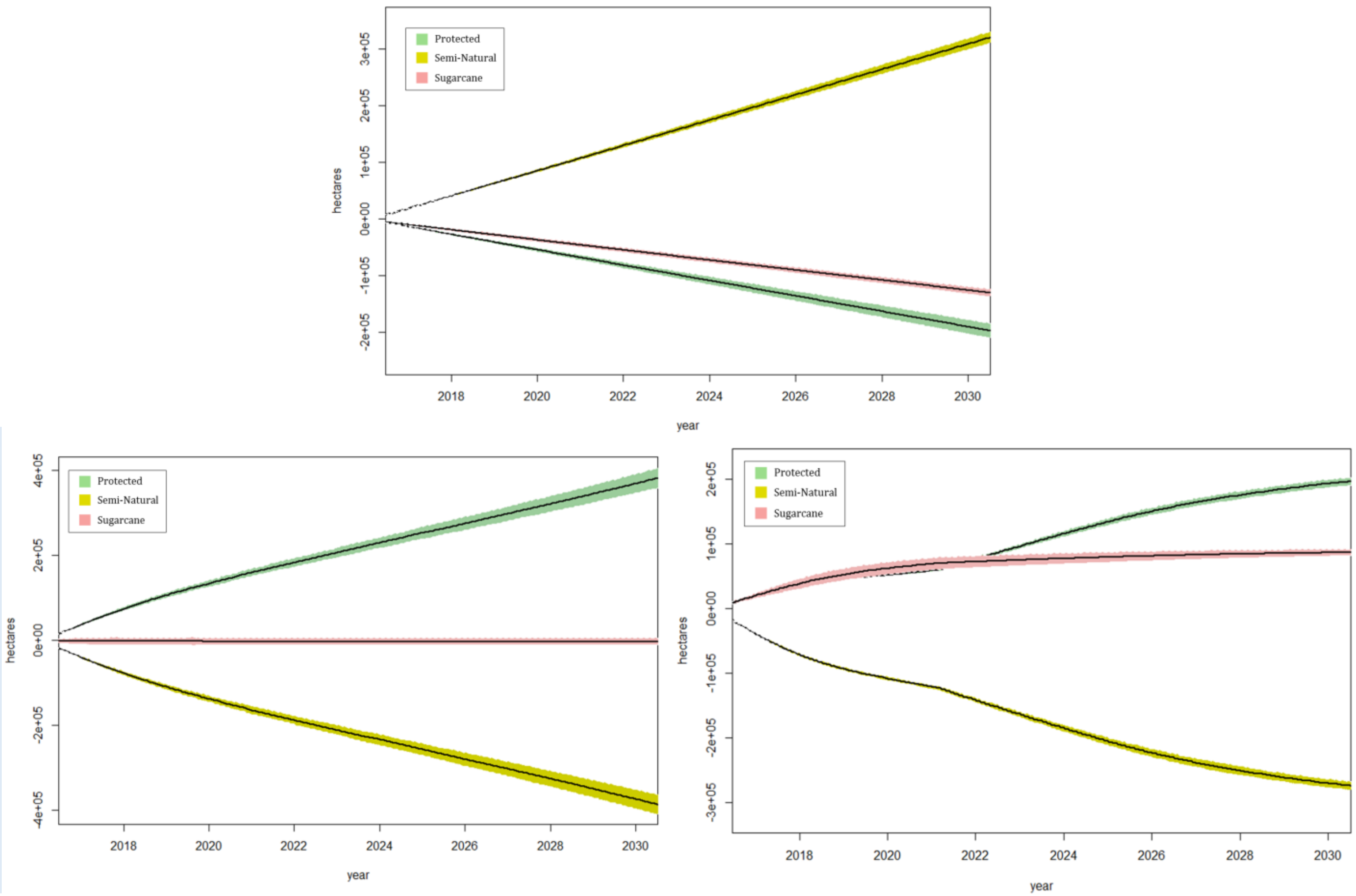

Figure 5.7: LUC trends for BAU (bottom left), LSP (bottom right) and LSH (top) for the three main land-uses analysed: protected areas (green), sugarcane (red) and semi-natural areas (yellow). Colour bands represent the standard error bands regarding all the runs computed for each indicator under every scenario. The black coloured lines show the mean values. *Source*: author.

While the BAU scenario shows positive trends for both biodiversity-related indicators and carbon sequestration – with steady state sugarcane production – the LSH scenario shows slightly positive biodiversity and carbon sequestration trends with decreasing sugarcane production. Regarding LSP, sugarcane production increases, while biodiversity remains stable and carbon sequestration decreases slightly. Nevertheless, increasing extinction debt values under this scenario may diminish net biodiversity in the long-term (see Discussion section).

Figure 5.8 shows the power and influence of protection and development governance forces on biodiversity at the landscape level. In particular, it shows the impact on biodiversity of two set of cases with different initial amount of protection forces driving land protection (*PG_p-agents)* and development forces driving land clearing for sugarcane production (*PG_d-agents)* (see Methods for a detailed explanation on *PG-agents*). The heatmap on top of Figure 5.8 shows biodiversity results with one single initial *PG_p-agent* and different initial number of *PG_d-agents*, while the bottom heatmap shows results for one single initial *PG_d-agent* and different initial number of *PG_p-agents*. The higher variability of biodiversity in the top heatmap compared to the bottom heatmap shows that, as expected, biodiversity in the Wet Tropics NRM Region increases considerably with stronger protection forces (i.e. higher number of *PG_p-agents*). In contrast, development forces (*PG_d-agents*) have a limited influence on biodiversity (bottom heatmap) even in those scenarios with strong development forces driving land clearing for agriculture (i.e. higher number of *PG_d-agents*). These results provide a baseline for governance discussion addressed in the next section.

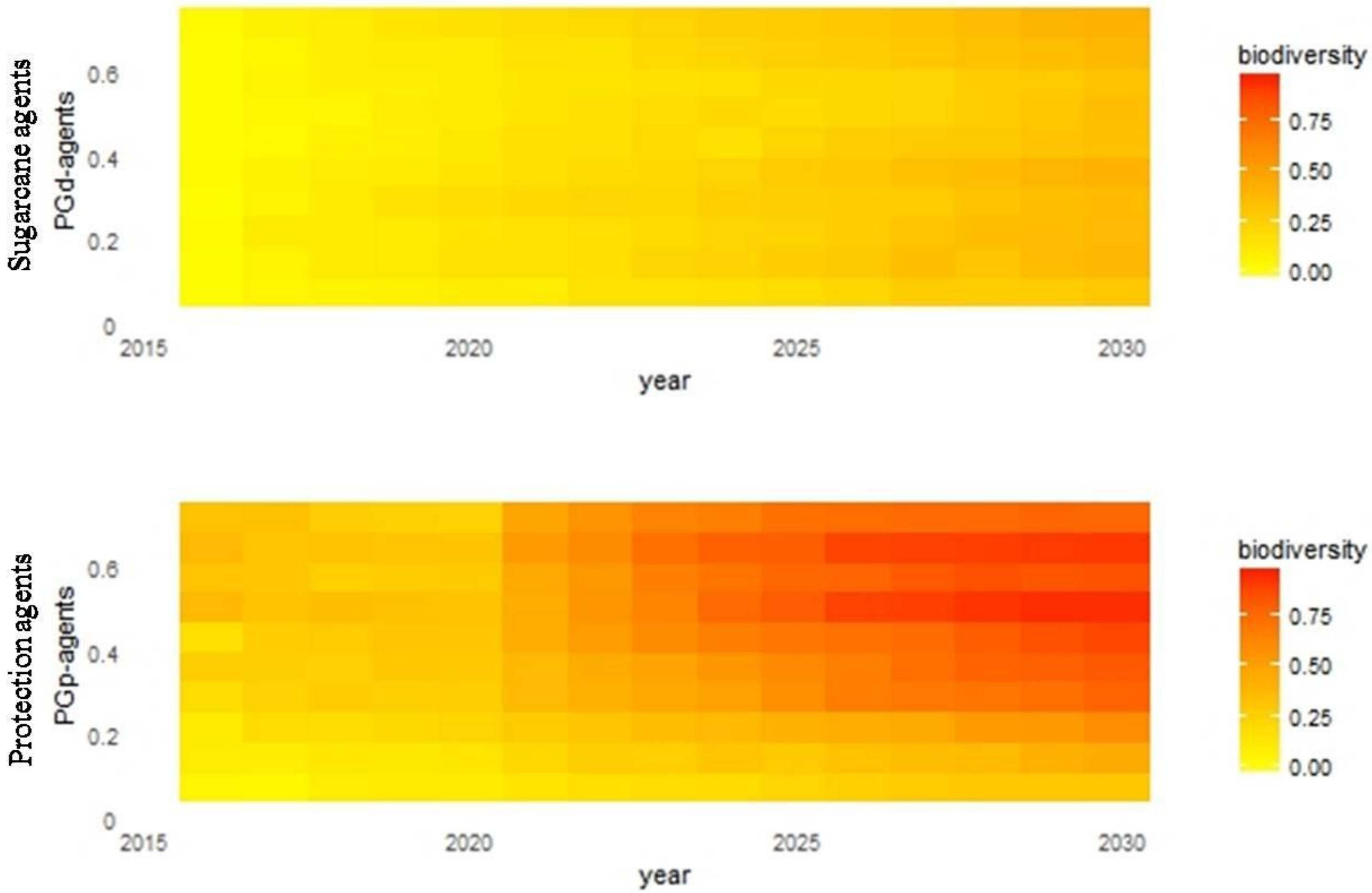

Figure 5.8: Impact of governance and political policy forces on biodiversity. *PG_p-agents* and *PG_p-agents* refer to conservation and development forces, respectively. The heatmap on top shows biodiversity variation over time considering the minimum number of initial *PG_d-agents* (i.e. five) for different initial *PG_p-agents* (i.e. values on Y-axis). The bottom heatmap shows biodiversity variation over time considering the minimum number of initial *PG_p-agents* (i.e. five) for different initial *PG_d-agents*. Only results for the LSP scenario are shown due to this scenario including development and protection forces competing for land. *Source*: author.

5.4 Discussion

5.4.1 What socio-economic and governance factors drive SES (un)sustainability in the Wet Tropics?

Under the framework considered, model outcomes show evidence that the current BAU scenario[15] in the Wet Tropics NRM Region is an atypical 'sustainable tropical island': providing food, conserving biodiversity and sequestering atmospheric carbon. These results are of more importance, considering that this tropical area – similar to other tropical regions – is managed under global, national and regional market economies seeking continuous economic growth, thus generally favouring land clearing for agriculture over conservation. Furthermore, traducing the outcome results from Figure 5.8 into a governance context, the heatmap on the top shows that the strength of the power of governance regimes currently needed to protect existing forests, maintain high biodiversity and limit land for development in the Wet Tropics NRM Region, is relatively low. Likewise, the heatmap on the bottom of Figure 5.8 shows that the strength of the power of governance driving land clearing for agriculture (i.e. sugarcane production) is not sufficiently strong to decrease biodiversity, even in those scenarios where development forces are considerably stronger than protection forces. Therefore, these results, together with the biodiversity outcomes obtained under BAU scenario (Figure 5.6), show that the Wet Tropics NRM Region would not need an excessive additional conservation governance power in order to maintain the current increasing biodiversity trends.

The positive biodiversity outcomes obtained under BAU could have been originated due to the combination of bottom-up and top-down conservation forces in the Wet Tropics NRM Region. While the latter force is driven by the institutionalization of biodiversity conservation through network policies and governance, the prior was originated in the 1970s through the growing bottom-up public knowledge and awareness of the

[15] Note that the BAU scenario in the case-study area is a 'partial LSP', where protected areas increase and agricultural production remains relatively stable over time.

environmental, social and economic significance of wilderness areas in the Wet Tropics (Burg, 2017). It was in that decade when the lack of substantial environmental movement that had dominated the North Queensland society since settlement in the 1860s started to change. Conservation groups, local citizens, and prominent national and international scientists initiated a battle, based on lobbying, direct action, mass mobilisation and political endorsements, against the economic forces driving land clearing. These were able to change public and government attitudes towards preserving the natural environment, thus shifting conservation strategies from a regional to a national – and at times a global – sphere (Brug, 2014). The Commonwealth became involved in the decision-making process, and the 'bottom-up' campaign culminated in the listing of the Wet Tropics Rainforests on the World Heritage Register in December 1988, which led to the beginning of a wide, strong and multilayer policy network for the protection of rainforest biodiversity. Currently, this multilayer policy network enables flexible, targeted responses to multiple and overlapping threats to biodiversity (Hill *et al.*, 2010; Hill *et al*, 2015bc). The result: currently almost fifty per cent of the Wet Tropics NRM Region is protected (DSITI, 2016), mainly rainforest, which helps protecting biodiversity and enhances the supply of multiple ES, such as global climate regulation, air quality regulation, and cyclone protection (Alamgir *et al.*, 2016).

Although the model results cannot provide with a more detailed analysis on the above-noted context – due to the abstract and conceptual nature of PG_p*-agents* (i.e. conservation-driving forces) – the stronger conservation forces in this case-study area, compared to land clearing forces, could have also been enhanced and reinforced by the following factors and processes: (1) *Socio-political* – timber harvesting from the tropical rainforests of north Queensland ceased following their inscription on the World Heritage List in 1988 (Vanclay, 1993). This helped re-electing a national government that took advantage of the above-noted bottom-up sentiment (i.e. environmental awareness) to make the logging ending process in the Wet Tropics a vote-winning argument nationally (Redfield, 1996). This decision was controversial in the sense that the State of Queensland, who was responsible for managing logging in state owned rainforests, argued that logging in this region was highly efficient[16]compared to tropical forestry elsewhere, as well as selective and intermittent – with low scale disturbances, similar to cyclones, to which the ecosystem is historically adapted (Nicholson, 1990). Regardless of whether imposition of preservation by the Federal Australian Government was positive or negative, conservation support by politicians, even if it was for their own political benefit, was an important factor enhancing the current environmentally sustainable context. (2) *Legal* – under the Australian Constitution, the national government can over-ride the states over matters tied to international treaties, such as the World Heritage convention. Although the management of the region itself corresponds to Queensland, the national government could stop environmentally unsustainable activities, such as the Wet Tropics logging. (3) *Environmental-scientific* – the region is the 2nd most irreplaceable globally in terms of its biota, including remnants of Gondwana that are not found elsewhere (Queensland Government, 2018). Because the Wet Tropics is a World Heritage Site (i.e. conservation hotspot), in contrast to most tropical areas located in developing countries, the prior is easier to justify and receive support with regards to conservation; (4) *Economic* – the tropical forests are around twenty times less productive of timber than temperate forests, where the latter provides the vast majority of the world's industrial wood (i.e. 75 percent) (FAO, 2004; Sedjo and Simpson, 1999). Moreover, the tourism industry in the Wet Tropics is making a much bigger contribution to the national economy (WTTC, 2017). Due to these factors, the influence of agriculture and timber industries in the region has decreased over the past decades. Besides this, as previously-noted, Australia is a rich, developed country, which means more funding for conservation purposes compared to tropical regions located in developing countries, which are more focused on solving poverty and other social issues (Redfield, 1996). It is important to note, however, that because timber is not regionally produced this does not mean that it is not imported. Thus, the Wet Tropics would be displacing the environmental impacts exerted by timber production elsewhere – this

[16] Utilization of timber resources was highly efficient in the Wet Tropics until inscription on the World Heritage List in 1988. For instance, while harvesting regulations caused nearly 150 tree species to be considered merchantable in the rest of Australia, operations in the Wet Tropics averaged less than 20 (Caufield, 1983). Furthermore, the State of Queensland utilised environmentally sensitive logging techniques, such as tree marking, careful location of roads, maintenance of riparian corridors, ban on logging operations during the Wet season (Poore, 1989).

issue, known as sustainability displacement, is addressed in Chapters 6 and 7 of this thesis; (5) *Governance* – public governance in Australia, compared to other countries in Southeast Asia, is currently doing better with regards to different indicators, such as corruption and poor governance (OECD, 2016; Sodhi *et al*., 2010). Governments showing low values for conventional indicators (e.g. corruption control, quality public services) are more likely to lead to spatial expansion of agriculture and Jevons paradox, while those governments accounting for the quality of environmental governance (e.g. reduce environmental stress, increase ecosystem vitality) generally enhance agricultural spatial contraction (Ceddia *et al*., 2014). Furthermore, public governance in Australia is more responsive to public opinion, which nowadays supports and requires a sustainable use of natural capital in the Wet Tropics. (6) *Geographical* – Australia has no spatial conflicts with neighbour countries (in terms of landscape management and protected area creation). Thus, the Queensland Government can manage the Wet Tropics NRM Region without having to deal with potential cross-national or international conflicts.

Besides the positive short- and medium-term results obtained for biodiversity, it is argued that we should be cautious at the time of making any assumptions in the long-term with regards to the situation in the Wet Tropics. This is due to the parallel results (to biodiversity) obtained for extinction debt (Figure 5.6). While the biodiversity figure (Figure 5.6) shows the variation of current (gross) biodiversity values affected by current LUC-driven habitat destruction/restoration processes, extinction debt (Figure 5.6) shows the future extinction of species due to events in the past, which occurs because of time delays between impacts on species and the species' ultimate disappearance. Thus, extinction debt provides key information about the equilibrium biodiversity in the Wet Tropics NRM Region, which refers to the future (long-term) net biodiversity values once extinction debt reaches zero and the system comes into equilibrium, without considering speciation (Tilman *et al*., 1999) – which is unlikely to occur in the model within the short timeframe modelled in the scenarios (2016-2030). The difference between the current (gross) biodiversity and the equilibrium (net) biodiversity is particularly important under the LSP scenario, where the short term positive-steady biodiversity results (Figure 5.6) could become negative in the long-term due to the increasing extinction debt (Figure 5.6). As a result of this, model outcomes presented in this chapter need to be considered with caution with regard to long-term conclusions.

5.4.2 What prevents sustainable development from happening in other tropical SES?

This section analyses the extent to which LSP or LSH approaches are suitable to manage tropical landscapes in terms of enhancing synergies among biodiversity conservation, climate change mitigation and food production. First, it is important to note that, following the results and context presented in the previous section about the Wet Tropics NRM Region, the current BAU context in this region cannot be compared to BAU scenarios from other tropical areas. The Wet Tropics is an atypical tropical region, with its own particular socio-economic, environmental, cultural and political context, where the current BAU scenario is sustainable from a SES perspective. Furthermore, the so-known BAU scenarios differ from place to place, where these undergo periods of nonlinear and abrupt changes that can end up in shifts to new regimes with markedly different economic and ecological characteristics (Muller et al., 2014); these regime shifts limit the predictability of land-system changes.

Due to all this, this section is also supported by data and information from the literature, instead of only the results obtained from the model presented, in order to provide a general picture of the question posed at the beginning of the section. Thus, a short answer to which approach is more sustainable (LSP or LSH) would be – it is context-dependent. Research has shown that it is difficult to meet different targets, such as the ones analysed in this chapter, under single LSP and LSH scenarios (Law *et al*., 2015; Ramankutty and Rhemtulla, 2013). Furthermore, the debate over the relative merits of LSP or LSH could be partly blurred by the differing spatial scales considered (Ekroos *et al*., 2016). Hence, no single spatial scale can efficiently segregate biodiversity protection and commodity production in multifunctional landscapes. In this regard, other approaches such as a mixture of LSH with LSP (Renwick and Schellhorn, 2016), multiple-scale land sparing (Ekroos *et al*., 2016), among others, have been proposed to go beyond a dichotomy that, for some scholars, is considered to be false (Renwick and Schellhorn, 2016).

Besides this currently ongoing debate, deforestation and biodiversity loss enhanced by land clearing processes under LSP approaches is a widely accepted fact that is diminishing the delivery of important ES for human wellbeing and the health of ecosystems (MEA, 2005). As previously analysed, stronger (than conservation) market-driven economic-development forces enhance unbalanced LSP contexts in tropical SES, i.e. land clearing for agriculture is prioritized over land protection and restoration (Hill *et al*., 2015). While this is a complex issue with multiple answers, the following arguments include potential factors that could be reinforcing the current unsustainable economic paradigm in most tropical regions: (1) funding for development is usually much higher than for conservation (Hill, 2015c). For instance, the leaders of the G20 nations gave a huge boost to the power of development regimes by promising to invest 60-70 trillion U.S. dollars on new infrastructure projects by the year 2030 (Hill, 2015c). There is no such investment for conservation, even less in developing countries where most tropical regions are located. (2) Profit-seeking businesses are generally given priority over conservation programmes. Brazil stands out in this respect, having one of the fastest increases in agricultural productivity in South America (together with Venezuela, Peru and Colombia) (Ceddia *et al*., 2014), and countries in Southeast Asia, such as Indonesia, where palm oil plantations keep growing at a considerable rate (Hill *et al*., 2013). (3) The amount of land protected in tropical regions does not normally reach the minimum 17% stated by the Aichi 2020 Targets (e.g. 14.7 per cent in Indonesia, 8 per cent in Malasya, 5.2 per cent in Panama, all in 2014) (World Bank, 2014). (4) Protected areas are usually located in very remote and isolated areas, thus reducing their positive impact on overall biodiversity (Palomo *et al*., 2013). Prioritizing protected area placement by proximity to active agricultural frontiers could make them more effective (Hill, 2015c), yet also more vulnerable regarding the potential effects of agriculture on the biodiversity of protected areas nearby. Hence, management of the surrounding territorial matrix of protected areas located next to intensive agricultural practices should not be ignored (Palomo *et al*., 2013). (5) Creating new protected areas through biodiversity offsetting should be considered as a valid action only when biodiversity benefits are additional to a baseline scenario (i.e. what would have happened without the impact) (Maron *et al*., 2015). Thus, using unsustainable BAU scenarios as baseline contexts will not diminish future threats to biodiversity (e.g. see for example the Cobre Panama copper-mine project in Maron *et al*. (2015)). (6) Public biodiversity discourses, rather than enhancing pro-conservation community sentiments, could be diminishing them (Hill *et al*., 2015a). This concept is based on the idea that society associates increases in protected areas with increasing pro-conservation community sentiments, thus leading to a public perception that more biodiversity is being protected(in the Wet Tropics), and thereby reducing public discourse about the risks of biodiversity loss elsewhere (i.e. rest of Queensland or Australia). In this regard, Hill *et al*. (2015a) argue that the creation of new protected areas in the Wet Tropics could be weakening protection forces elsewhere in Australia, especially in Queensland – where only 7.92 per cent of land is currently protected (far below the 17 per cent stated in the Aichi Biodiversity Target 11). Thus, there is a need for further research exploring the environmental and socio-economic relationships between (geographically) distant SES, instead of solely performing single coupled SES analyses. The so-called telecoupling framework provides a conceptual platform in this regard, which could help providing more realistic results with regards to different sustainability issues (this framework is used later on in Chapter 6 as a proposal for further research).

5.5 Conclusions

The current BAU context in the Wet Tropics NRM Region is the most sustainable scenario from a SES perspective. This is mainly because conservation forces are stronger than economic forces, which has its origin in different processes related to socio-political, legal, environmental-scientific, economic, governance and geographical factors. This is an outstanding achievement for a tropical region, considering that the same market and profit-seeking forces driving unsustainable land clearing and development in other tropical regions (WTMA, 2017) are also present in the Wet Tropics.

Deciding about LSP or LSH is not an either-or proposition, since a mixture of sharing and sparing will be need to meet conservation goals in a world with a growing demand for different ES. Furthermore, considering that SESs are complex, dynamic, and nonlinear systems – where BAU scenarios likely change from one place to the other – the potential to extrapolate specific contexts and solutions among tropical SESs is low.

Thus, each geographic context and set of stakeholders will need to explore alternative sustainable SES systems, based on their own local and regional socio-cultural, economic and environmental contexts – without abandoning the importance of meeting global goals (e.g. SDGs). The situation in the tropics will be even more complex, where land tenure and governance are not clearly defined, biodiversity and conservation values are heterogeneous across the landscape, and business models normally enhance land intensification over sharing (Kremen, 2015). Nonetheless, the LSP versus LSH framework has the potential to meet multiple goals that, when integrated within spatially explicit models, can be used to explore sustainable solutions for complex social-ecological systems.

Chapter 6:

Sustainable development: Why is it not delivering on its promises?

> "*The economy is a wholly owned subsidiary of the environment, not the other way around*"
>
> – Gaylord Nelson (American politician, 2002, p. 85)

The aim of this chapter is to address the research objectives posed in section 1.2 (Chapter 1) based on the results obtained from Chapters 3-5. Thus, to analyse which combination of socio-economic and governance factors lead to the (un)sustainability of SES, as well as the role of power relations between economic and conservation forces in this regard – where the debt-sustainability relationship is particularly examined. Prior to addressing these objectives, it is important to analyse the relationships among the models developed in this thesis (see Chapters 3-5) and the way in which these are combined under the conceptual framework presented (see Chapter 2). This first step is a necessary preparatory work to synthesize and compare the modelling results obtained, thus allowing for an in-depth understanding as a whole of thesis's results. The first section of this chapter (6.1) provides an integrated modelling platform, which justifies and describes the relationships among models by applying the conceptual framework to each model. This analysis serves as a basis to examine, in section 6.3, the factors driving (un)sustainability outcomes in complex SES, based on the results derived from this thesis and further literature review.

6.1 Ontology matching: Integrating Agent-Based Models to explore sustainability in complex social-ecological systems

The conceptual framework presented in Chapter 2 was specifically constructed to address the research objectives posed in Chapter 1. This framework was applied in Chapters 3-5 to explore SES (un)sustainability, as defined in this PhD thesis, in complex SES. Each of the models presented in Chapters 3-5 share the same conceptual framework.

From a modelling perspective, the framework is related to the concept of an *ontology*. An *ontology* is an inventory of the kinds of entities that exist in a domain (e.g. framework), their important properties and the salient relationships that can hold between them (Benjamin *et al*., 1995). In computer science, ontologies are defined by Gruber (1993) as formal, explicit representations of shared conceptualizations. Thus, in a short and simplified way, an *ontology* may refer to the shared structure, entities and processes between different models. Because the three models presented in this thesis were built under the same conceptual framework, they share most of the main elements from this framework. As a result, these models share, and are built under, one single main *ontology*, i.e. an explicit modelling representation of the (shared) conceptual framework, including its entities and processes. At the same time, because each model computes its own particular processes, relationships and sub-entities – based on specific research questions and context (see the UML class and activity diagrams for each model in Appendix A, B, C) – each model has its own *sub-ontology*, which form part of the thesis's overall *ontology* (Figure 6.1).

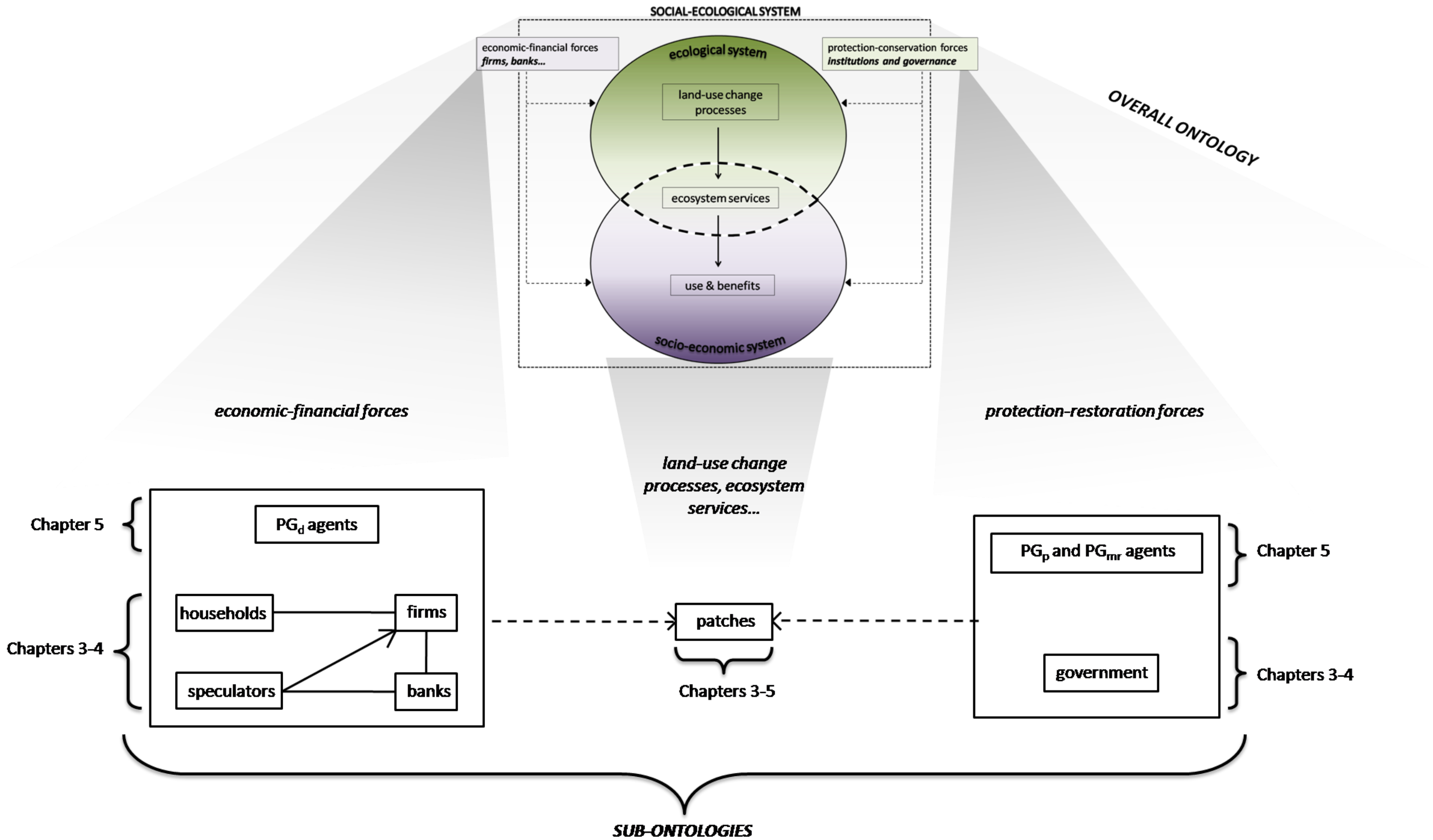

Figure 6.1: Overall ontology of the thesis. The top part of the figure shows the conceptual framework, while the bottom part shows the application of the framework to each model. In particular, the sub-ontologies (bottom part) refer to the specific application and integration of framework elements within each particular model. For instance, protection-restoration forces are represented by the government in Chapters 3 and 4, and by the PG$_p$ and PG$_{mr}$ agents in Chapter 5. The same case is applied to economic-financial forces, while patches represent, in all models, the nexus included in the conceptual framework regarding land-use change processes–ecosystem services–use &benefits. The specific applications of the framework to each model (i.e. sub-ontologies) are obtained, and can be integrated within, a main ontology, which represents the overall modelling application of the thesis' conceptual framework. *Source*: author.

Besides describing the thesis's ontology (Figure 6.1), it is also important to structure, in detail, the differences and similarities between the models' sub-ontologies. The concept of *interoperability* refers to the conditions under which a formal correspondence between two different (sub-)ontologies can be established (Polhill and Salt, 2017). In this context, (sub-)ontology comparison, or interoperability, can be seen as matching ontological elements between at least two differing (sub-)ontologies. Table 6.1 displays the model interoperability of this thesis; the tables showing which main elements (i.e. entities, processes) from the conceptual framework are *explicitly* or *implicitly* present in the models, thus providing a direct comparison between model sub-ontologies.

Tables 6.1: Model interoperability. Both tables show the application of the main conceptual framework elements in each model. Check and cross marks show the *explicit* and *implicit* simulation of framework elements in the models, respectively; while check marks refer to those elements that are specifically modelled, cross marks refer to those elements implied in model entities and processes, but not directly modelled. Footnote numbers highlight key elements that need further clarification – analysed in detail below. Tables 1 and 2 therefore serve as a platform to compare the presence, and form, of framework elements among models. *Source*: author

	SES		**economic-development forces**			**protection-conservation forces (government)**				
	ecological system	socio-economic system	banks	firms	speculators	protected area policies	land restoration policies	technology & innovation policies	land clearing policies	speculation policies
Chapter 3 (**conceptual**)	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Chapter 4 (**Indonesia**)	✓	✓	✓	✓	✗ [3]	✓	✓	✓	✓	✗ [3]
Chapter 5 (**Wet Trop.**)	✓	✓ [1]	✗ [1]	✗ [2]	✗ [3]	✓	✓	✗ [4]	✓	✗ [3]

	land-use change processes			**ecosystem services & biodiversity**					**use & benefits**
	protection	restoration	land clearing	carbon sequestration	carbon emissions	food production	biodiversity	others	households
Chapter 3 (**conceptual**)	✓	✓	✓	✗	✗	✗	✗	✓ [6]	✓
Chapter 4 (**Indonesia**)	✓	✓	✓	✗ [5]	✓	✓	✓	✗	✓
Chapter 5 (**Wet Trop.**)	✓	✓	✓	✓	✗ [5]	✓	✓	✗	✓ [7]

The following points describe, in detail, those framework elements highlighted in Table 6.1 (see footnote numbers in Table 6.1) – note that the order followed below is based on the left-to-right footnote order from Table 6.1. In particular, each point below analyses the implicit or explicit integration of the framework element in the corresponding model(s):

1. **Implicit modelling of debt, including banks (Chapter 5)**. The explicitly modelled economic system in the Wet Tropics case-study is, like Chapters 3 and 4, a debt-based economic system. However, in Chapter 5, borrowing is not explicitly performed by economic forces driving land clearing for agriculture (i.e. PG_d-agents[17]), but rather implicit to the economic system itself. Thus, although sugarcane production is a debt-driven activity in the Wet Tropics (Katter, 2014), the model neither explicitly tracks, nor accounts for, debt dynamics and stocks. The reason for this is twofold; (1) compared to Indonesia – where both debt and palm oil production increase over time, with a high correlation and dependency between both (see Chapter 4) – sugarcane production in the Wet Tropics has remained relatively stable over recent years (see Chapter 5). In the Wet Tropics, credit facilities (debt) are mainly used to maintain the current small production of sugarcane and prevent the crash of the industry (Katter, 2014). This makes the debt-production relationship in the Wet Tropics relatively weak. Therefore, tracking debt dynamics, and their relationship to production and environmental sustainability, is not as relevant in the Wet Tropics as it is in Indonesia. As a result, further allocation of credit facilities to the sugarcane industry in the model, or testing different debt scenarios, would not (in my opinion) have an effect on production. In addition to this, (2) the model presented in Chapter 5 uses Hill *et al.*'s (2015a) system dynamic model as a basis, where debt is not explicitly modelled.

2. **Implicit modelling of firms (Chapter 5).** Firms are not explicitly represented by an additional type of agent in Chapter 5, but rather implicit to PG_d-agents (i.e.

[17] PG-agents (i.e. PG_d, PG_p and PG_{mr}) is an abbreviation, taken from Hill *et al.* (2015) and used throughout Chapter 5, to refer to those forces driving land clearing, land protection, or land restoration, respectively, e.g. governments, banks, farmers, communities. For example, PG_d-agents (i.e. economic forces) in Chapter 5 would be similar to banks and firms, with the help of household demand, in Chapters 3 and 4, since these are economic forces driving production.

development forces driving land clearing for sugarcane production). Thus, PG_d-agents represent firms (i.e. sugar mills, farmers and all actors involved in the production of sugar), among other entities, that drive the actual production, management and selling of sugar. Furthermore, and similar to the previous point, Hill *et al*. (2015a) do not model firms independently, but rather include their effects within development (economic) forces.

3. **Implicit modelling of speculation and speculation policies (Chapters 4 and 5).** Speculation was integrated in the conceptual model (Chapter 3) as a representative indicator of financial assets, in order to explore the extent to which the latter affects the economic-environmental (de)coupling. The main influence of speculation on the sustainability of SES is through prices, since it enhances inflationary price scenarios that affect economic and environmental indicators ultimately. Speculation was not explicitly modelled in the case study-based models (Chapters 4 and 5) because it was already implicit to the (changes in trends from) crude palm oil and sugarcane prices, respectively – which were based on historic data.

4. **Implicit modelling of technological development and efficiency (Chapter 5).** The Wet Tropics model (Chapter 5) does not explicitly simulate technological development and efficiency, yet this is implicit in the production of sugarcane in the model. Similar to the above-noted points (1 and 2), Hill *et al*.'s (2015a) model, which was used as a basis to build the Wet Tropics model, does not include technology as a parameter. Besides this, the impact of technological efficiency on the Wet Tropics SES sustainability is not modelled because the socio-economic, environmental and governance context in this region is more suitable to address one other more relevant factor for the SES sustainability: the power (im)balance between government and economic-development forces, represented by conservation-production conflicts (Hill *et al*., 2015a). The model was, therefore, used to explore whether power relations and conflicts are affecting different sustainability indicators, instead of focusing on the role of technology in this regard – which had already been studied in Chapters 3 and 4. As a consequence, technology was solely (implicitly) integrated as part of the forces driving sugarcane production (i.e. PG_d-agents).

5. **Selecting different climate change mitigation indicators for each case-study model: emissions and carbon sequestration (Chapters 4 and 5).** The explicitly modelled climate change mitigation indicators in Chapter 4 and 5 are carbon (CO_2) emissions and carbon sequestration, respectively. The reason for selecting different indicators in each model was based on the different composition and distribution of the land-use matrix in each case-study. Thus, land clearing for agriculture – and, therefore, carbon emissions – is very limited in the Wet Tropics (Chapter 5), while the amount of carbon sequestrated is considerably higher there due to the large protected forested area (around 50% of the region) (DSITI, 2016). Due to this, carbon sequestration values were considered more suitable to track changes in carbon dynamics and stocks for the Wet Tropics. In contrast, the amount of carbon emitted through the currently increasing land clearing for palm oil production in Indonesia (Chapter 4) is considerably higher than the amount of carbon sequestered from protected forests (Paltseva *et al.*, 2016; Pearson *et al.*, 2017). Thus, tracking carbon emissions was considered a more suitable indicator regarding changes in carbon stocks and dynamics for the Indonesian case-study.

6. **Using the stock of a conceptual regenerating natural resource as an environmental indicator (Chapter 3).** The model presented in Chapter 3 is a conceptual model. Therefore, in contrast to Chapters 4 and 5, ES and biodiversity are not used as environmental indicators, but only the stock of a conceptual regenerating natural resource type (e.g. timber). This decision was made as this chapter is mainly focused on studying the impact of debt-based economies on environmental sustainability (i.e. debt-sustainability relationship), as well as the role of government intervention in this regard. The aim was therefore to model a simple regenerating resource, representing a form of 'natural capital', which could be used as a general environment indicator affected by different debt-driven economic activities. This environmental indicator was, thereafter, disaggregated into different and more specific environmental indicators for the case-study models, i.e. Chapters 4 and 5.

7. **Implicit modelling of households (Chapter 5).** Households, in Chapter 5, are implicitly modelled as the demand driving sugarcane production in the Wet Tropics, instead of explicitly modelled as an additional type of agent. This is based on Hill *et al*. (2015a) not including any specific entity for households or demand; thus, the most suitable solution for disaggregating such elements into the ABM was to implicitly model household demand[18] within the sugarcane production process.

The aim of this section was to analyse the links (both similarities and differences) among the models presented in Chapters 3-5, as well as their integration within the conceptual framework (Chapter 2) of this thesis. The integrated modelling 'platform' presented in this section, including the results obtained from the previous chapters, is used in section 6.3 to address the research objectives and question posed in Chapter 1.

6.2 Sustainable development: Why is it not delivering on its promises? Insights from a multidisciplinary social-ecological systems perspective

The three models developed in this thesis studied the interactions and feedbacks between an economic system, representing the current debt-based economic system, and the environment. The current capitalist economic system cannot survive without growth. Growth is financed through the accumulation of more and more debt, which forces the society to generate future growth to repay the debt with interest, thus creating a negative reinforcing debt-growth cycle (Daly, 2010). Furthermore, the expected expansion of the global human population (possibly reaching 10 billion people by 2100), together with the increased standard of living and GDP (which increased nearly 60-fold in real terms from 1820 to 2003), creates the perfect argument to continue with the current economic growth paradigm (IPCC, 2014; MA, 2005; United Nations Secretariat, 2012). The fundamental problem of the economic system is that it rewards short-term production and consumption at the expense of stewardship of natural capital

[18] Note that, in Chapter 5, 'household demand' over sugarcane represents the total global demand on this good – not only coming from households – yet the term 'household' is maintained for the sake of similarity with the rest of the models.

(see WBCSD (2017)), which is necessary for human wellbeing in the long-term (MA, 2005). Thus, achieving sustainable development will not be possible without understanding those factors affecting natural capital – and, therefore, our long-term wellbeing – and how can we integrate natural capital in the economic system, including policy, decision-making (WBCSD, 2018a,b).

In this section, I ask: based on the results obtained in Chapters 3-5 – under the conceptual framework used in this thesis and the modelling approach selected – *which socio-economic and governance factors are hindering sustainable development of SES, and what is the role of economic and conservation powers in this regard?* This section analyses, in detail, the conclusions derived from Chapters 3-5 (section 6.2.1). This is followed by an analysis that develops, to a certain degree, further the results obtained to outline the need of an alternative pathway to achieve sustainable development in the long-term (section 6.2.2).

6.2.1 Power (im)balances and dynamics between economic and conservation forces in capitalist economic systems

Scholars argue that, if economic growth is not absolutely decoupled from environmental pressure, the systems that support life on this planet are going to collapse at some point in the near future (Smith *et al.*, 2011). Under the conceptual framework and modelling approach used in this thesis, results show that there is a power imbalance, under business as usual (BAU) scenarios, between financial-development forces driving economic growth and conservation forces driving environmental sustainability. In this thesis, it is argued that the current economic system is not by definition (i.e. *per se*) environmentally unsustainable. Rather, the particular behaviour of economic agents in the system, including the inappropriate use of economic assets and technological development, as well as the weak (conservation) governance power, which are key factors currently enhancing SES unsustainability. The results show that, if both economic and environmental sustainability are to be achieved[19], there is a need to (i)

[19] Besides the interconnectedness between the three dimensions of sustainability (i.e. environmental, economic and social), this thesis solely focuses on the first two (see Chapter).

enhance a shift in the current economic-capitalist forces to support environmental sustainability, as well as to (ii) strengthen the role of the State in protecting the environment from the rough edges of the market economy.

The following sub-sections (6.2.1.1-6.2.1.3) analyse, in detail, those key factors driving (SES) sustainability, based on the conclusions derived from Chapters 3-5.

The role of economic-financial institutions

The models in Chapters 3-5 shown that debt is a key driver of SES (un)sustainability. Yet, debt-driven economic systems do not impose a growth imperative *per se,* but the behaviour of entities and agents in the system, as well as their decisions and relationships with regards to the environment, show a tendency to increase SES unsustainability. Regarding the particular use that the private sector and other actors make of debt (i.e. credit facilities), international policy makers and non-governmental actors have become increasingly concerned about the entry of speculators into the system (UNCTAD, 2011; Cox, 1976). Speculators contribute to the creation of volatile, artificial and difficult-to-predict speculative markets, which distort commodity prices by creating excess price volatility. Thus, debt is not used for production-oriented activities – which benefit society by producing goods and creating job positions – but rather for speculative processes that only provide profits to speculators. Thus, those periods when speculation follows positive increasing trends show a high disconnection of the economy with regards to the environment; in contrast, those periods where artificial speculative markets are absent show contexts where economic elements are better coupled to the environment. It is clear, from this results, that speculation globally could be an important financial actor affecting economic trends and, therefore, environmental sustainability (Galaz *et al*., 2015). There is a need to reduce the level of speculation in the system in order to align changes in prices to supply-demand processes, as well as to the state of natural resources and other environmental indicators. As a result, this could help in obtaining more realistic values regarding the decoupling processes between gross domestic product (GDP) and environmental pressures.

The role of technological development and efficiency

The model from Chapter 3 shows that, under current BAU scenarios, technological efficiency increases natural resource extraction rate compared to resources' growth rates, which consequently results in system collapses. On the other hand, results from Chapter 4 suggest that improving production efficiency through technological development in existing (palm oil) plantations could enhance SES sustainability. Based on these 'opposing' results, it is argued that technological development itself is not the factor enhancing the decoupling between GDP and environmental pressures, i.e. technological develop can be both positive and negative for the environment depending on the context. Rather, it is the *rate* at which technological and efficiency is developed what states whether technological efficiency is negative or positive for the environment (and our society). Thus, high technological efficiency rates enhance negative reinforcing cycles (i.e. mismatches) between natural resource extraction and resource growth rates, as well as between extraction rates and the capacity (velocity) of the government to implement conservation policies. These results align with Gunderson and Pritchard (2002) with regards to the fact that policy deficiency in sustainably managing natural resources has not kept pace with the speed of (technological) changes that alter the processes in SES. In the model (Chapter 3), the implementation rate of conservation policies is not high enough to counterbalance the exponentially increasing rates of technological efficiency. Therefore, a slower, but steady, increase of technological development, as occurs in Chapter 3, could help decoupling GDP from environmental pressures and, thus, prevent future system collapses.

The lesson to be learned here is that technology is a necessary element for sustainable development and decoupling economic growth from environmental pressures (see, for example, the success at decoupling economic growth from SO_2 pollution due to technological development, described in Chapter 2). However, technology needs to be cautiously funded, managed and evaluated. In this regard, the Precautionary Principle (PP), a strategy that operates on the frontiers of scientific knowledge and decision-making, aims to cope with possible risks where scientific understanding is yet incomplete (De Sadeleer, 2010). Basically, this is a principle to determine how safe is safe enough – regarding technological development – thus urging institutions to act or abstain from action, in case of uncertainty (De Sadeleer, 2010). While in most cases it should encourage the delay – or even the abandonment – of activities suspected of having serious impacts on the environment, conversely it should accelerate the adoption of decisions intended to ensure better environmental protection (De Sadeleer, 2010). In short, PP encourages a new relationship of policy, decision-making with science, where the latter is consulted less for the knowledge which it has to offer than for the doubts and concerns which it is able to highlight. Although sometimes considered a broad and fuzzy concept, the PP is quickly developing into one of the foundations of environmental protection in the EU; for example, the European Court of Justice (ECJ) ruling against Spain in Marismas de Santoña (a protected wetland), for not having sufficient protected wetlands for certain migratory species of birds (Zsuzsanna *et al.*, 2004). Spanish authorities justified the destruction of a valuable ornithological site by the fact that no reduction in the number of protected birds had been observed in such area. To counter this argument, the ECJ applied the context of uncertainty (i.e. PP) resulting from the fact that destruction of a natural habitat does not necessarily translate into an immediate decline in its animal populations. Thus, the ECJ considered that bird populations could be actually declining due to current development-economic activities, although such a decline was obvious from up-to-date indicators. This delay in the extinction of species and habitat loss, highly important regarding the PP, is related to the concept of extinction debt (Tilman *et al.*, 1994), which is modelled and analysed in Chapter 5 of this thesis.

The application of the PP is important for SES sustainability since technological development has tended to accelerate over the last decades (Modis, 2002). The problem lies on the support for technology and innovation as a source of ever increasing efficiency, thus justifying economic growth (Lafforgue, 2008). Yet, relying on technological development to maintain the current continuous exponential growth is contested by the uncertainty and unpredictable nature of technological innovation (Lafforgue, 2008). In fact, technology could be giving a “false” stability (Allison and Hobbs, 2004), since technological advances usually focus on single variable interventions without considering their impacts on other parts of the systems (Helpman, 1998). This has been described as human propensity to focus on “single cause-and-effect solutions” (means-ends logic designed to solve a particular problem) (Westley *et al.*, 2002). Furthermore, another sustainability problem arises from the Jevons Paradox;

this idea establishes that increases in efficiency of resource use are usually outpaced by the rate at which consumption of those resources increases (Jevons, 1865). Thus, further technological efficiency does not always mean less consumption.

It is clear that technological development is a key element for achieving sustainable development, yet it has to be addressed with caution. In particular, technology can contribute both by increasing the efficiency of the use of the extracted resources – including key aspects from the circular economy (Pearce and Turner, 1990), such a sustainable and fair resource distribution (Brears, 2018; Daly, 1992;) or efficient waste management (Stanhel, 2016) – and by increasing the efficiency with which resources are extracted (McKinsey Global Institute, 2017). In fact, based on the resilience theory, we will need novel technology to effectively prevent the whole socio-economic system crossing critical (resource extraction) thresholds, and to help sustainably managing the whole system (related to resource use and distribution) (Allison and Hobbs, 2004). Note that, the technological development and efficiency modelled in Chapters 3 and 4 of this thesis is related to extraction and production processes, rather than resource waste, use or distribution.

Besides the potential benefits of technology with respect to environmental sustainability, it is necessary that we change the dependency of the current system on exponential technological advances that cause the degradation and unsustainable use of natural resources through economic growth. However, it has long been known governments' reliance on technology to solve environmental problems, due to their reluctance to make the social and political changes necessary to reduce growth in production and consumption (Sharon, 1994). Moreover, as it is known from many other complex past civilizations, innovation does not always prevent socioeconomic collapse (Blanton and Tainter, 1990; Smil and Diamond, 2005). The following question, therefore, remains: can exponential advances in technology occur without causing major socio-political, economic and environmental changes, and without a rethink of political and social priorities? It seems that, because technology is not independent of society either in its shaping or its effects, we will probably need further solutions and social-economic changes, apart from further technological development, if the aim is to enhance a more sustainable and circular economy.

The role of bottom-up and top-down conservation forces

Results from Chapters 3-5 show the need for stronger government driven conservation strategies and the implementation of policies that protect the environment. Weak environmental governance, above all in tropical SES, fails at counter-balancing economic powers that continuously seek for short-term profits. At the same time, this situation is reinforced by weak public environmental awareness and bottom-up conservation forces.

Hardin's (1968) tragedy of the commons highlighted the difficulties of sustainably managing common goods in shared-resource systems. Ostrom (2007, 2009) argued that actors tend to improve in this regard only when resources are scarce. Interestingly, results from Chapter 5 show that the situation in the Wet Tropics of Queensland, Australia, is different, where a system with abundant resources can still be sustainably managed under certain circumstances. Based on these results, the Wet Tropics is an atypical '(environmentally) sustainable tropical island': providing food, conserving biodiversity and sequestering atmospheric carbon. These positive results regarding current (and future) environmental sustainability originated from the rise of diverse bottom-up conservation forces in the past. Driven by different social actors (e.g. conservation groups, NGOs, local communities, citizens, scientists), such bottom-up conservation forces were the starting point, in the 70s and 80s, for taking conservation more seriously at the governance and policy-making level in the Wet Tropics and North-East of Queensland (Burg, 2017). In particular, grassroots mobilisation/movements – e.g. lobbying, direct action, mass mobilisation, political endorsements – were the consequence of the growing public knowledge and awareness of the environmental significance of wilderness areas in this region (Burg, 2017). The importance of these past events, as shown by Chapter 5 results, lies in the fact that this tropical area was managed under the same global, national and regional market economic forces seeking continuous economic growth that are present in other tropical areas.

Under the resilience theory, small-scale disturbances – or what some have referred to as "shock therapy" (Gunderson and Holling, 2002) – are encouraged (Biggs *et al*. 2012), as they contribute to the adaptive capacity of the system and its ability to innovate.

These small-scale disturbances can promote the use of alternative pathways to transfer information throughout the system (Schweinberger *et al.*, 2014) by seeking opportunities in the way sub-systems interact on different time scales (Hector *et al*. 2010; Isbell *et al.*, 2009). Small-scale disturbances, such as the bottom-up conservation forces in the Wet Tropics, encourage the participation of diverse actors that can contribute to the system in a variety of ways, beyond those of the traditional key actors and leaders. Encouragement and creation of these small-scale disturbances can increase the adaptive capacity of the system when it is confronted with large-scale disturbances (Gunderson and Holling, 2002). By studying small-scale disturbances within a system, one can gain some information about their frequency and effect on social and political organizations.

Results obtained from Chapter 5 show a good example in this regard. In short, bottom-up environmental awareness and dynamics led to the current top-down, multilayer conservation force in the Wet Tropics, which enables flexible, targeted responses to multiple and overlapping threats to the environment (Hill *et al*., 2010, 2015bc). Thus, the positive results obtained – from an environmental perspective – under the BAU scenarios for the Wet Tropics case-study (Chapter 5) are due to the combination of strong top-down (conservation) governance forces with long-term established bottom-up enterprises in the Wet Tropics. As a result, 50 per cent of the Wet Tropics is currently protected, a considerably larger area than the area occupied by the main industry – sugarcane production – which covers around 8 per cent of the region (DSITI, 2016). Little land clearing for sugarcane plantations occurs in this region, with sugarcane production values remaining steady stable over time.

Besides the Wet Tropics, certain studies point to some specific factors as central to continued deforestation, such as a clientelistic system (a social order depending upon political patronage) dependent on resource extraction supporting elites (Fleischman *et al*., 2014). In this regard, results obtained in Chapters 3 and 4 show similar results, where the role of financial powers, such as banks, is key to understand deforestation rates in Indonesia and other tropical SES. For instance, private companies, operating in various sectors in Southeast Asia – palm oil, pulp and paper, rubber or timber – receive loans in millions of dollars from overseas banks, including the US (Bank of America), Europe (Credit Suisse) and within Asia (Forest and Finance, 2016). These loans fund land clearing. As an example, in 2015, at least $43 billion in credits were loaned to companies linked to deforestation and forest burning in Southeast Asia alone (Forest and Finance, 2016). The idea behind these data is that forces driving protection in most tropical regions are not sufficiently strong to halt land clearing driven by debt-based economic forces (Hill *et al*., 2015a). Therefore, there is a need for stronger conservation governance, to help compensate with the negative environmental impacts exerted by the generally stronger financial powers. In fact, good conservation governance has proved to be successful in reducing deforestation and increasing protected forests in some tropical areas, such as the Amazon (Soares-Filho *et al*., 2006). Yet, high corruption and low public governance quality could be hindering SES sustainability through ineffective funding allocation for forest, wildlife and natural resource conservation (Sodhi *et al*., 2007). As discussed in Chapter 4, the problem here is the political difficulty of implementing policies that, indirectly, reduce the power of influential financial institutions. Thus, governments are usually not free to invest or create new institutions that could help enhance SES sustainability, but must take account of the influence of industries and other interest groups (Abel *et al*., 2006) – see section 6.2 below for a more detail analysis on this issue.

In short, the models presented in this thesis (Chapters 3-5) have shown that a combination of strong and multilayer bottom-up and top-down forces can enhance environmental sustainability in complex SES. This is a challenging scenario to be achieved in most SES, considering that the Wet Tropics case shows an atypical context and possesses unique characteristics unlikely to be found elsewhere. The reality is that the current economic system – not to say our wellbeing – depends on the state of natural capital, and the problem comes from humanity depleting natural capital faster than the Earth can replenish it (Costanza *et al*., 1997). Hence, there is a need for a paradigm shift if the aim is to integrate the value – not only economic – of natural capital in our daily economic activities and the economic system as a whole. Furthermore, this becomes even more important if one integrates public and common goods into the equation. Under the current economic system, there is no agreed-upon framework, neither at the national- or global-level, nor at the local and business levels, that provides effective solutions for the so-called *negative externalities*, or market failures (Barnes, 2006). To what extent is the current capitalist system, and all the actors involved, prepared and able to deal with this issue? The next section builds upon the results from this thesis to (i) disaggregate the problem of natural capital and negative externalities, and (ii) provide potential, general pathways to start moving towards a more (long-term) sustainable future for our society.

6.2.2 Needing a 'bounded economy'

The blindness of the free market economy

Results from this thesis show that some degree of government intervention is necessary to counterbalance the negative effects of the economy on the environment. In the real word, this is particularly relevant when environmental public goods – e.g. clean air, soil water, landscapes – or common goods (i.e. 'common pool resources') – e.g. fish stocks, timber – are involved. Regardless of the difficult categorization between public goods and commons (Felipe-Lucia *et al.*, 2015; OECD, 2001), the current capitalist system is characterized by commonly leaving environmental assets as open access; this permits corporations to gain control over environmental assets, thus enhancing their over-use and degradation (Rittenberg and Tregarthen, 2012). As discussed in Chapter 2, the failure to internalize the value of environmental assets (into the system) is known as negative externalities or market failures. Such *market blindness* takes place whenever, and wherever, economic activities produce influences that affect realities external to the logic of the system, given that these influences are invisible to both the economic activities and the system itself. Hence, the capitalist system fails to adequately value and manage the commons (Barnes, 2006), while the failure of other political systems such as communism, for instance, has been traced to the desire to put too many assets into the commons (Costanza, 2007).

For a long time, economists were assured that there was no need to be concerned about negative externalities, due to their triviality with respect to the economic system and the high wealth created by the latter (Barnes, 2006). However, the lack of balance of our economic system soon became too obvious to be ignored. The pattern that is clear in the literature is that, whenever and wherever market blindness exists, an official body must regulate the market; and the agent responsible to intervene is usually the State (Barnes, 2006). Interestingly, the notion of setting limits and correcting market behaviour is not new at all, and it can be traced back to the writings of Adam Smith; Smith (1776) argued that the State was the regulator of the proper institutional environment, and focused on facilitating the virtues of the market (Smith, 1776). Even more radical approaches, such as Zero Growth or a Steady State Economy (Daly, 1973, 1977, 2008), have the assumption that government policies are the only approach possible to help internalizing externalities.

Economists and public officials have historically been aware of these deficiencies of the market-based economic model, as well as of the options available to mitigate them. Among the variety of options available, the following are the most well-known approaches to internalize externalities (DeNyse, 2000): *standards* – such as emission standards, a legal limit on how much pollutant a firm can emit; market-based regulations, including *taxes* – on goods and services, which enhances the revenue of governments and reduces the quantity of goods produced and consumed (e.g. 'polluter pays principle', where the economic agents causing environmental harm carry the economic costs of the negative externalities they create (Gomez-Baggethun and Perez, 2011)– *tradable emissions permits* – permits allocated to firms to generate emissions – and *subsidies* – i.e. a negative tax, where the government pays the seller and buyer, or both, to produce and consume more of a particular good or service (e.g. 'steward earns principle', where the beneficiaries of ecosystem services should economically compensate the stewards that maintain or protect the services from which they benefit(i.e. Payments for Ecosystem Services (PES)) (Wunder, 2005); *trade policies* – policies affecting the market by changing the way firms and individuals of different countries interact with one another; *auctions* – where the government, which controls the rights of part of the natural capital found within a country, acts as a participant in the market for these natural resources and sell them to the highest bidder. Yet not as widely-used as the above-noted mechanisms, the *inclusion of natural capital in GDP* is another option to correct the deficiency of capitalism of not assigning any value to natural capital and living systems – which are the basis of human capital. GDP calculation has the drawbacks with respect to the environment (Hawken *et al*., 1999): first, the value of clean air, clean water and extensive forests are not included as a part of the GDP – yet people feel worse off if they live in a country that shows low values of these indicators. Second, man-made capital is depreciated, but not natural capital – a country that exhausts its man-made capital without replacing it clearly grows poorer, while one that exhausts its fish stocks or minerals may appear to grow richer based on conventional measures. Finally, the cost of cleaning up environmental damage is recorded as an addition to GDP – the environmental loss caused by the damage in the first place is not recorded.

Criticisms with regards to the use of these instrumental mechanisms have increased in recent years. Is taxation a good tool for preserving the commons? In the first instance, this idea makes sense – if pollution is free, there will be no incentive to reduce it; while if it is taxed, there will be. Yet, as Barnes (2006) suggests, the 'devil' is in the details: who sets the taxes? How quickly can they act, and to whom are they accountable? Where does the money go? The problem is that, eventually – taking carbon taxes as an example – the prices of gasoline, natural gas, or electricity could rise as soon as governments set high taxes, thus affecting both the private sector and households. Therefore, a scenario where governments set policies that are, in the short-term, against their own national economic interest, is highly improbable. As discussed in the next section, it is important to understand the severe limitations that the State faces to implement policies regarding the internalization of externalities – this can be applied also to conservation policies and governance in general.

The powerless or unwillingness of the State to constrain corporations

In the models presented in Chapters 3-5, environmental conservation is carried out by conservation forces (see Figure 6.1 in this chapter), which are mostly represented by *governments*. In theory, one could argue that governments – which are not driven by profit maximization – constitute the perfect entity to represent the commons sector of the economy; governments would be, therefore, in a suitable position to prioritize the natural capital for the common good, rather than supporting the private interests of corporations. However, history has taught us that governments are not necessarily the ideal counterweight to corporations, since they have, too often, become the representatives of private interests, thus leaving the commons underrepresented, undervalued and underprotected (Costanza, 2007). The difficulty of states to successfully enhance long-term productive and sustainable use of natural resource systems has been present in literature for a long time now (Ostrom, 1990). In this regard, the State usually faces fierce corporate resistance whenever it tries to exercise its powers, thus regulatory agencies are too often highly influenced by the industry (Barnes, 2006). Corporations exert pressures on governments through different investments and lobbying, to prevent governments from implementing measures that go against the private interests of firms. Such investments are usually considerably lower than the potential revenues to be gained by corporations, if the latter are successful in halting governments' plans. As an example, the timber industry in the U.S. spent $8 million in campaign contributions in 2012 to preserve a logging road subsidy worth $458 million; the return on their investment was, eventually, 5,725 percent (Barnes, 2006). Furthermore, the pressure on governments may be also based on their dependency, as well as that of economic growth, on corporations. For instance, one could imagine that the CEOs of several major companies decided they want a government bailout worth many millions; they could tell Congress that, without receiving X millions, their companies would not survive, therefore, affecting the socio-economic system as a result of the consequent collapse. If the government agrees – which is often the case – each company would gain the amount received by the government divided by the number of corporations involved in the deal. In the process, the average taxpayer would only lose a few (monetary) units through taxes. There is, therefore, no point, from an economic perspective, for ordinary citizens to 'fight' for their rights in this particular context – since the loss for citizens is much lower compared to the gains for corporations. This sort of scenario is only one of (many) that enhance corporations' decision-power in the system, in comparison to that from governments.

Another factor enhancing the power and influence of corporations, in the current system, is their greater expertise, compared to government experts and policy-makers, in their own sector. Corporations are in a position of power, thus being able to exert pressure and convince governments to follow their recommendations on certain matters. This, together with corruption, is a key aspect enhancing a power imbalance between the State and the private sector under capitalist systems (Hopkin and Rodriguez-Pose, 2017). As an example, Spain has moved from being the world leader in solar power in 2010, to being overtaken by many other countries. The so-called "sun tax", a national law approved by the Spanish Government in 2015, taxed solar installations focused on self-consumption – i.e. installations that just produce for their own use and do not feed into the grid – among other measures (Abaco Advisers, 2017). The lack of support to solar energy by the government was partially due to the Spanish electricity market being in deficit, with its running costs exceeding the sale of power (Abaco Advisers, 2017). Although the reasons underlying this deficit are up for debate, experts argue that the government was acting in the interest of big business, rather than those who were pushing for sustainable self-consumption and combating climate change (Abaco Advisers, 2017). In particular, accusations are directed at key ministers sitting on the boards of the electricity companies, therefore, supporting energy consumption from oil and gas companies instead of investing in the renewable sector. Moreover, the fact that political parties usually only rule for periods of four or five years – while corporations have long history – have a wide, continued experienced in the field – creates the context where the latter can 'convince' the government to support plans for the benefit of corporations. This situation does not help with implementing long-term policies benefiting commons preservation. Finally, another scenario showing the power imbalance between corporations and the State is represented by so-called regulatory capture (Barnes, 2006). Details vary, but the plot is always the same. A new agency is created to regulate an industry that is harming public goods or commons. At first, the agency acts independently, but over time it starts making decisions to benefit the industry. Reformers who originally staffed the agency are replaced by people who either worked in the industry previously, or hope to do so after a period of employment in government. As a result, an agency that was, in the first instance, created to protect the public sector, ends up benefiting the private interests of corporations.

Politicians and corporations usually have a symbiotic relationship, where politicians need revenues and corporations want favours (Barnes, 2006). Based on the above-noted factors, the two main actors of the economic system (i.e. corporations and governments[20]) are not able to help internalizing externalities. Changing the nature of

[20] Note that those government interventions (in the economy) modelled in the models of this thesis represent all conservation forces driving, as a whole, environmental protection (not only governmental).

the private sector and re-building corporations – that is, to make them driven by something other than profit – is an unlikely scenario, since corporations are created to make money for their stakeholders and, as a society, we want them to make money, employ people and pay taxes. Similarly, to liberate government from corporations – not just momentarily, but long-lastingly – is another unlikely scenario, considering that corporations have powerful tools to counter this potential scenario, as previously discussed. As a consequence, the current capitalist, market-based economic system lacks institutions and contexts that incentivise the preservation of the commons, or charge corporations for degrading nature. At this stage, therefore, an obvious question arises, with no simple, straight forward answer: how could the commons be sustainably managed, under an economic system that supports continuous and environmentally unsustainable economic growth, with neither structural/internal incentives, nor effective external ones (i.e. governmental), to reduce the loss of public goods on which our wellbeing and society depend?

Laying the foundation stone: The need of a 'commons sector':

The answer to the above-noted question is complex, probably requiring systemic changes (both structural and superficial) at different scales. Traditional conservation approaches, whilst having achieved numerous objectives in terms of protection of rare species and habitats, are powerless to reverse or stabilize the metabolic patterns of a global economy that encourages ever-increasing demands on natural capital stocks, ES, and biodiversity (Gomez-Baggethun and Perez, 2011; Guo *et al.*, 2010). Although the state of the environment would undoubtedly be worse if conservation strategies had not been in place, traditional conservation has so far failed to reverse the loss of natural capital (Armsworth *et al.*, 2007).The conservation movement has thereby failed to act upon the economic and sociopolitical drivers of change that are at the root of many present environmental problems (MEA, 2005; Steffen *et al.*, 2004).

A first step to reverse this situation could involve the recognition of the importance of the commons as a separate and distinct sector of the economy. Thus, commons need to be valued and 'propertized', yet not privatized, through entities managing such common property rights. These entities could be constituted by *hybrid* institutions that are neither

(i) influenced by corporations, nor (ii) afraid of voters or politicians. Barnes (2006) argued that the U.S. Federal Reserve Board constitutes one example of a hybrid institution. Created in 1913 to manage money supply, this entity is a mix of a corporation – whose stock is owned by member banks – and a public institution – since the seven members of its board of governors are appointed by the president and confirmed by the Senate. The measures they carry out (e.g. to raise interest rates) can be detrimental for both corporations and voters; no politician would want to make such decisions, yet thanks to this entity none has to. Thus, entities with a similar nature to the Federal Reserve Board's could be able to make tough economic decisions to preserve the commons – such as reducing deforestation or raising energy prices – without committing political suicide. Despite the obvious success of the Federal Reserve Board – reflected by the fact that, nowadays, almost all countries use it – there is a lack of examples of similar entities managing environmental commons.

There are obvious questions and issues revolving around the creation of hybrid institutions, e.g. how to avoid corruption of the board members? Who would set the prices for the commons (in cases where it was possible to account and measure them), and which instruments would be used in this regard? Is economic valuation the only type of valuation that should be used at the time of establishing commons properties? These are difficult questions to answer. Also, who should own the commons? Markets, as previously discussed, cannot be, since they only seek to maximize profits and have no structural mechanisms to protect environmental assets. Among the options left, governments are political entities with a short time horizon and a high dependency and influence regarding the private sector. The suggested hybrid entities, by contrast, could represent institutions, similar to current common property trusts (Libecap, 2008), which establish property rights for the commons, without privatizing them. This sort of trusts, forming a 'commons sector', would have long time horizons and a legal responsibility to future generations. There should be also variety among such entities; the commons sector should not be a monoculture like the corporate sector, but rather each institution should be appropriate to its particular (natural) asset and site. As a result of turning commons into common (not corporate) property, what are now unpriced externalities could become property rights under accountable management. Thus, if corporations had to pollute to carry out their activities, they would have to buy the rights from a commons trust. Moreover, the importance of establishing properties for the commons lies on taxes, laws and regulations – as mechanisms to internalize externalities – being easily rescinded or weakened by corporations. Property rights, in contrast, tend to endure, as do institutions that own them (Barnes, 2006). In an ideal society, this approach would be applied to all our common natural and social capital assets, while leaving private property intact.

Although the current society and economic system is miles away from the described scenario, this theoretical commons sector would not have to be built from scratch. There is already an enormous potential base just waiting to be claimed, both in terms of commons and active trusts – or institutions that could take the role of the above-described trusts. Examples of trusts nowadays include the Alaska permanent fund (APFC, 2018) – a trust set up by the State of Alaska to manage royalty payments for oil and gas extraction – and the REDD (United Nations, 2018) – the United Nations Framework Convention on Climate Change policies for reducing emissions from deforestation and forest degradation. Most part of the work undertaken by organizations such as REDD is based on research performed within the fields of natural capital and ES, including Payments for Ecosystem Services (PES) (Farley and Costanza, 2010; Wunder, 2005).PES seeks to internalize the environmental externalities of human actions, ascribing monetary value to ES and helping decision-makers to recognize the real value of the loss of ES (Bellver-Domingo *et al*., 2016). Although rudimentary forms of PES have been in place for many decades[21], it was not until the mid-1990s that Costa Rica set up, for the first time in history, PES schemes at the national scale (Pagiola, 2008). More recently, schemes for international PES have been promoted, such as the Clean Development Mechanisms (launched in the 6th Conference of the Parties (COP) of the Kyoto protocol) and the above-noted REDD and REDD+ programmes.

Nevertheless, the increasing implementation of PES schemes contrasts with concerns and challenges related to the capacity of PES policies to internalize externalities and enhance the provisioning of multiple ES. The REDD+ framework will be used as an

[21] In the 1930s, the US Government promoted payments for farmers that adopted measures against soil erosion (Jacobs, 2008).

example here, yet these and other similar issues can also be found in PES schemes applied to other sectors, such water-related PES (e.g. Zanella *et al*., 2014) and agricultural PES (e.g. Vorlaufer *et al*., 2017). As previously mentioned, REDD offers incentives for developing countries to preserve and enhance forests to offset the growth in global GHG emissions. One of the first challenges for PES, as a potential mechanism to help internalizing externalities, is related to funding. As demonstrated by the models in Chapters 3-5 of this thesis, economic forces driving land clearing for production are normally stronger than conservation forces driving protection. For example, oil palm agriculture, which has become a major drive of tropical deforestation, is currently more profitable than preserving forests for carbon credits or PES (Butler *et al*., 2009). Although it is expected that 200 billion Euros will be transferred through PES schemes by 2020 (GIZ, 2016), there is still a need to rise investments for PES in order to be able to compete with the agricultural sector or any other profitable human activity (Butler *et al*., 2009).

A second challenge with regards to PES is related to the fact that implementing PES schemes can be a ‘messy’ process (Ishihara *et al*., 2017). Actors are forced to assemble and adapt their actions by combining and considering schemes and institutions embedded at different scales (e.g. locally and nationally). For example, PES schemes established at national levels may have to be reshaped by local actors (e.g. farmers) to fit their local context, or local institutions may have to fit the national PES scheme into the locally dominant ‘institutional logic’ (Ishihara *et al*., 2017). Using the REDD as an example, scholars have criticized the incapacity of REDD policies to adapt to domestic mechanisms, as well as difficulties of regional and national processes to reconcile with the evolving demands of the international REDD infrastructure (Evans, 2017).Thus, the success of REDD within individual countries may also depend on its ability to be flexible enough to traverse complex and shifting domestic processes (Evans, 2017). An exemplifying case is given by Norway and Indonesia. In 2010, Norway and Indonesia signed a US$1 billion deal aimed at reducing deforestation in Indonesia, where the Central Kalimantan region was selected as the REDD pilot province (Lang, 2010a). However, eight years after, the agreement has not made much difference to the rate of deforestation. One of the main problems was that the REDD measures were implemented by Indonesia in already degraded and deforested areas, while massive expansion of tree and oil palm plantations still continued in remaining natural forest (Lang, 2010a). Furthermore, because the agreement would take up to two years to be implemented, oil palm companies reacted and took advantage of the situation by increasing oil palm production and, therefore, deforestation (Lang, 2017). The aim of palm oil corporations was, therefore, to create as many plantations as possible until the REDD program was officially implemented.

This example shows that preserving and protecting natural capital is not only a matter of allocating funding for conservation, but also dealing with the complex task of integrating and adapting regional, national and international processes and targets. This argument can be also applied to other sustainability issues, such as the implementation of the Sustainable Development Goals (SDGs) (discussed in Chapter 6), which will require the integration and adaptation of international goals with commitments and mechanisms at lower scales.

Besides this, other challenges to be addressed by PES if externalities are to be internalized include problems with ES accounting, measurement and distant links between ES. With regards to the latter, recent research shows that ecosystem assessments often overlook what is described as “distant, diffuse and delayed” impacts (Pascual *et al*., 2017). As an example, protection of a coral reef from fishing may lead to more fishing in neighbouring sites. This idea states that global SES, including their actors and ES, are interrelated at different scales, which means that these burdens must be better recognized and incorporated in ecosystem assessments (Pascual *et al*., 2017). Chapter 6 in this thesis proposes further research focused on studying the socio-economic and environmental relationships of distant SES, as well their impacts on ES trade-offs. As for the challenge of measuring natural capital, while some ES, such as carbon, are considerably easy to measure, others are more difficult to account for. Biodiversity, yet not specifically an ES, can be considered an example in this regard (Gaston and Spicer, 2004). In fact, complex biological resources have largely been known to be a greater challenge to the design of sustainable institutions to manage them (Becker and Ostrom, 1995). Finally, the economic valuation of ES through ESP schemes has also been criticized by some scholars. While the environmental economics approach to PES tries to force ecosystem services into the market model, with an emphasis on efficiency, the ecological economics approach seeks to adapt economic institutions to the physical characteristics of ecosystem services, prioritizing ecological sustainability (Farley and Costanza, 2010). Decision-making processes are normally based upon instrumental values with regards to nature (values pertaining to the economic value of an ES), and these are, indeed, critical to conservation (Kai *et al*., 2016). However, thinking only in these terms may miss a fundamental basis of concern for nature, based on intrinsic values and the so-known ‘relational values’ (Kai *et al*., 2016). While intrinsic values pertain to the value inherent in an object, relational values pertain to all manner of relationships between people and nature, including relationships that are between people but involve nature (e.g., a relationship of impact via pollution, which is mediated by a watershed). Thus, it is important that policy-making and PES mechanisms also integrate these other values and are not solely driven by economic approaches.

Coming back to the creation of property rights for managing the commons, PES can be considered a first step forward with regards to the internalization of externalities (Bellver-Domingo *et al*., 2016). Nevertheless, the creation of entities that are, in principle, against the current main economic paradigm – whose objective is to propertize but not privatize the commons – would probably meet fierce resistance from corporations. It is, therefore, clear that overcoming that stage would require some sort of leadership from certain actors, focused on helping these entities to successfully fulfill their mission. For instance, and using the American context as an example, Barnes (2006) argues that anticorporate forces have come to power once or twice per century, e.g. the eras of Jackson and Lincoln in the 19th Century; those of Theodore and Franklin Roosevelt in the 20thCentury. In resilience theory, effective leadership recognizes the structure of complex systems and thus understands how the system functions at different levels (Fath *et al*., 2015). Thus, it is the responsibility of the leadership to ensure that actors within the system have the resources and guidance to continue their trajectory in a way that is beneficial to the rest of the system (support positive feedbacks). Moreover, leadership can also be originated and represented through bottom-up forces, as discussed previously in this chapter and Chapter 5. The Wet Tropics thereby moved from an almost non-existent number of protected areas and environmental movement since settlement in the 1860s, to the current 50 per cent of the region protected, including a strong conservation institutionalization and high environmental awareness and responsibility among politicians, citizens, local community and the tourist industry (Burg, 2017). While the 21st Century equivalent leaders are still emerging, Barnes (2006) argues that the job of the current society is to be ready when they come. Readiness, in his view, constitutes three things, namely to (i) to assign common property rights to trustworthy entities; and (ii) recognize that the duration of any anticorporate ascendancy will be brief, thus such small windows of opportunity have to be effectively used to build institutions that outlast it. By carrying out these premises, establishing a commons sector that balances the corporate sector would be closer than before. This new sector would offset the corporate sector's negative externalities with positive externalities of comparable magnitude. The result could be a balanced economy that gives us the best of both sectors and the worst of neither.

6.3 Thesis contributions and future work

6.3.1 Theoretical contributions

Due to the interdisciplinary nature of this thesis, results and conclusions derived from the models contributed to the literature across several related disciplines. This thesis provides new insights regarding the transdisciplinary fields of sustainability science (Bettencourt and Kaur, 2011; Clark and Dickson, 2003; Kates, 2011; Kates *et al*., 2001; NRC, 1999; Raven, 2002) and ecological economics (Costanza, 1989; Costanza *et al*. 1997; Daly and Farley, 2004; Turner *et al*., 1993). Regarding ecological (macro)economics, one of the challenges facing this discipline is based on studying the relationship between macroeconomic elements of the system and environmental processes that operate at lower scales. This was addressed through Chapter 3 and, to a greater extent, Chapter 4. Both interrogated the effect of monetary debt, as a macroeconomic element, on ecological dynamics at sub-national and regional scales. Furthermore, the three models (Chapters 3-5) explored also those socio-economic and governance factors that are necessary to enhance economic and environmental sustainability[22] in complex SES, which is a key question within both sustainability science and ecological economics. Moreover, the models explored ES trade-offs and bundles, from both a spatially-explicit (Chapter 5) and temporal dynamic (Chapters 4-5) perspective, giving a broad of what specific areas, and when, ES trade-offs are more likely to arise.

The knowledge gaps within ecological economics and sustainability science addressed have also implications for research on coupled human and natural systems (Alberti *et al*. 2011), coupled social-ecological systems (Walker *et al*. 2004), and coupled human-environment systems (Turner *et al*. 2003). The three models simulate the interactions and dynamics among key social and economic actors in the current economic system, and with the environment. The models, therefore, embrace the complexities of dynamic SES. Building SES models, or coupled human-natural systems models, are necessary not only to inspire interdisciplinary research on key issues for sustainability, but also to facilitate communication among scholars interested in nested systems as well as single disciplines (McGinnis and Ostrom, 2014). Besides this, the trade-offs between human well-being and the natural environment modelled in Chapters 3-5 also have implications for ecosystem services research (Daily, 1997; MEA, 2005, TEEB, 2010); in particular regarding ES trade-offs (Bennett, Peterson and Gordon, 2009) in Chapters 4 and 5, and spatially explicit mapping of ES trade-offs (Turkelboom *et al*., 2018) in Chapter 5.

6.3.2 Modelling contributions

The Agent-Based Land-Use Modelling (ABLUM) community highlights specific challenges regarding ABM for the coming years. One of them relates to the spatial representation of SES through ABM (Filatova *et al*., 2013). Several key publications from the turn of the Century highlighted pioneering work on spatial ABM for SES (Gimblett, 2001; Grimm, 1999; Kohler, 2000). The importance of spatial representation

[22] The three dimensions (or pillars) of sustainability (including the social) are interconnected, thus directly, or indirectly, affecting each other at multiple temporal and spatial scales. However, as pointed out in Chapter 6, this thesis focuses on the environmental and economic dimensions by only modelling environmental and economic indicators. Although the thesis' social implications are not addressed in the models (e.g. social inequality, poverty), further model versions could integrate social indicators considering that the social-economic and environmental systems are already modelled.

of SES lies in the fact that SES almost always operate in a highly variable spatial environment (Filatova *et al*., 2013). In this regard, the empirical and spatially-explicit model presented in Chapter 5 extends the literature on spatial simulation of SES through ABM. Furthermore, Chapter 5 also contributes to the need to build ABMs that link emergent properties to macroscopic patterns of ABMs, or other modelling tools (O'Sullivan, 2016). Examples of links between ABMs and other modelling techniques include system dynamic models (Miller *et al*., 2014) and hybrid tools (Geertman *et al*., 2009), among others. In particular, hybrid models consist of integrating different modelling techniques to help reconciling the advantages, and reduce the weaknesses, of the modelling approaches selected (O'Sullivan, 2016). The model presented in Chapter 5, therefore, contributes to the demand for further hybrid models – in this case by linking Bayesian Belief Networks (BBN) and Geographic Information Systems (GIS) with ABM. It is important to note that, although the use of BBNs for modelling land-use change (LUC) is not new (see Bacon, Cain and Howard, 2002; Lynam*et al*., 2002), examples including the incorporation of BBNs into spatial ABMs are scarce (Kocabas and Dragicevic, 2013; Lei *et al*., 2005). The integration of BBNs into the ABM from Chapter 5 helped addressing uncertainties related to agents' decision-making, based on probabilistic approaches that followed expert opinion and GIS data (see also Perez-Minana, 2016). Finally, Chapter 5 contributes to the need to develop further empirically and theoretically grounded models (i.e. 'mid-level' models) (O'Sullivan, 2016). Mid-level models are characterized by the appropriate balance between empirically-rich and theoretically-grounded approaches. Thus, these models are realistic enough to represent the salient dynamics in a particular system, without incorporating many elements or dynamics that affect the ability of the modeller to interpret the model (O'Sullivan, 2016). The model in Chapter 5 uses empirical and spatially explicit data under a well-known conceptual framework in land-use and landscape science (Land Sharing [LSH] versus Land Sparing [LSP]) (Phalan *et al*. 2011, 2013; Melo *et al*. 2013; Ramankutty and Rhemtulla 2013; Scariot 2013, von Wehrden *et al*. 2014).

With regards the model presented in Chapter 3, yet not as empirically rich as the model in Chapter 5, it uses the economic "Circuit Theory" framework (Graziani, 1990) as a basis, as well as Steve Keen's (2009, 2010) credit-based macroeconomic models. Chapter 3 also shows an example of an ABM built upon disaggregating macroscopic patterns and modelling processes from another top-down, system dynamic model, i.e. Steve Keen´s models (2009, 2010). Thus, Keen's models are adapted to the characteristic bottom-up, emergent properties driven by heterogeneous agents in ABM.

Appendix B, pp. 73-74, show further knowledge gaps and further research within the ABM community that is not related to this thesis.

6.3.3 Further research: Using the telecoupling framework to explore the impact of power relations in SES (un)sustainability.

The contributions and new insights provided by this thesis were conducted within individual systems (i.e. SES). Although processes and data from external (to the modelled) systems were integrated and considered for some aspects (e.g. global debt for the Indonesian SES, Chapter 4), little attention was paid to the impacts and distant interactions with other SES. This is an important aspect to consider, where one needs to be aware of the 'panacea trap' (Ostrom *et al*., 2007) that states that it is essential to avoid providing simplified pictures of SES and recommend cure-all solutions. In this regard, interactions among distant systems are increasingly widespread and influential, with profound implications for sustainability (Liu *et al*., 2013). Similarly, misunderstanding or misperceiving the spatial scale of populations, resources or systems is a well-known issue in natural resource management (Wilson *et al*., 1999). In this regard, the conceptual framework of *telecoupling* uses an integrated approach to study the socio-economic and environmental interactions among coupled human and natural systems over distance (Eakin and Wehbe, 2009; Liu *et al*., 2011, 2013), i.e. SES that are spatially/geographically separated from each other. I will use the soy trade between the regions of Rio Grande do Sul (Brazil) and Lower Saxony (Germany) to illustrate the phenomenon of (interregional) telecoupling – and the interactions among distant entities, such as states. Trade in soy between the two regions has increased dramatically over the past decade, with significant economic, social, and ecological implications for both regions (Lenschow *et al*., 2016). Intensive meat production in north-western Germany is highly dependent on Brazilian soy as a basis for livestock feed, while Brazilian soy producers rely on European export markets – particularly Germany – for their product (Grenz *et al*., 2007). This connection affects multiple ES and other sustainability indicators in both regions; for instance, demand over livestock and soy production are associated with soil and water degradation, loss of biodiversity, habitat destruction (Fearnside, 2001; Kessler et a., 2007), among other environmental impacts.

The concept of telecoupling is a logical extension of research on coupled social-ecological systems (Walker *et al*. 2004) or coupled human-environment systems (Turner *et al*. 2003, Moran 2010). Telecoupling is also a unifying concept that builds upon previous concepts such as teleconnection, globalization, and world systems theory (Gotts, 2007; Dreher *et al*., 2008; Hornborg *et al*. 2007), which have largely been limited to single disciplines (see Adger *et al*. 2009, Seto *et al*. 2012). Because telecoupled systems are hierarchically structured, the framework takes a multilevel analytic approach, which can be described in the following manner (see Figure 1 in Liu *et al*. (2013) for a diagram): at the telecoupled level, the framework includes interrelated (yet geographically separated) coupled SES that are connected through socio-economic and environmental flows. Within each SES, there are three interrelated components: agents, causes, and effects. Each of these components, at the same time, includes many elements or dimensions, e.g. different types of agents (individuals, households, corporations). Furthermore, there are cross-level interactions among coupled SES, e.g. agents from one specific SES facilitate flows among other SES, and flows among coupled SES produce effects in each SES.

The telecoupling framework, as with other frameworks, aligns particularly well with certain specific modelling approaches more than others. In this regard, the telecoupling framework could not only be linked to ABM, but also help improve ABMs. For instance, ABM is widely used for research in land use change and coupled human and natural systems (Filatova *et al*., 2013; Polhill *et al*., 2011). However, most agents in the ABM literature are simulated within coupled SES – even though ABM has the potential for doing so in multiple SES. The telecoupling framework, in this regard, calls for explicitly incorporating interactions among agents in distant coupled systems, or telecoupled agents, in shaping land use change and dynamics of coupled systems. A telecoupling perspective could help develop more realistic models and future scenarios through ABM, and more accurate forecasting to reflect an increasingly telecoupled world.

Besides the application of the telecoupling framework through ABM, the interest in this conceptual framework lies on its capacity to study the role of power relations for global sustainability. In particular, the interdisciplinary, multilevel and integrative nature of the telecoupling concept aligns with the demand for further research addressing cross-scale links between financial and ecological systems (Soranno *et al*., 2014). More specifically, a sound understanding of how key macroeconomic issues are entangled with environmental shifts and destructive feedbacks at lower levels is missing (Klitgaard and Krall, 2012). Further models are, therefore, needed to study the extent to which global financial markets and actors (e.g. financial powers, demand for commodities, price volatility) drive land use- and ecosystem changes at local scales in complex and indirect ways (Galaz*et al*., 2015; Lambin and Meyfroidt, 2011). An example of local-scale ecological impacts from broader-scale financial processes includes producers affected by increasing global price volatilities, such as farmers and farmer organizations (Chavas *et al*., 2014; Morales, 2017). These global-scale changes prevent the latter from making long-term investments that would increase agricultural productivity and production, and from transferring to more sustainable production methods (FAO and OECD, 2011). Related to this, Chapter 3 showed the great influence of speculation on price volatility and, therefore, SES sustainability. Another example includes those communities depending on forests for their food security, which are often vulnerable to higher food prices. Rapid increases in prices tend to force these communities to extract more resources from forests for sale in local markets, with direct impact on forest species and ecosystems (Chavas *et al*., 2014).

In short, further work with regard to this thesis would consist of studying, and modelling, the socio-economic and environmental relationships between (geographically) distant SES under the telecoupling framework. In particular, this approach could be used to study the impact on SES sustainability of power relations and (im)balances between debt-driven economic forces and environmental conservation forces. The following section describes a potential research case involving Norway and Brazil under a telecoupled framework.

6.4 Conclusions

The models presented in this thesis have shown the complex and interconnected relationships between the economy and natural systems, and between economic and conservation forces, in coupled social-ecological systems. Furthermore, modelling results have shown the advantages of using integrative, holistic and interdisciplinary approaches for addressing sustainability issues, as well as the need to consider and integrate the specific socio-economic, cultural, political and environmental contexts under case-study-based approaches.

The research question and three objectives posed in section 1.2 (Chapter 1) have been answered and addressed throughout the thesis, based on the results obtained from the three models presented in Chapters 3-5. In particular, answering such research question involved addressing (simultaneously) the three research objectives proposed at the beginning of the thesis; namely (i) to study what *combinations of socio-economic and governance factors* drive SES (un)sustainability in complex SES; (ii) to investigate the *relationship between (monetary) debt and SES (un)sustainability*; and (iii) to examine the effect of *economic and conservation powers (forces)* on SES (un)sustainability. All the three objectives were addressed through the three models presented in Chapters 3-5, whose main results and conclusions have been presented and explained in section 6.2.1 of this chapter. Furthermore, addressing these objectives served as a basis to answer the main research question of the thesis: *what hinders sustainable development under the current capitalist economic system, and is there a built-in bias towards environmental unsustainability?* Under the particular conceptual framework, modelling approach and case-studies selected, the answer to the second part of the question is *yes*. Results have shown the ever-increasing use of natural resources under credit-based, market-driven SESs, where the economy does not protect the natural capital on which it depends. This is the case for most business as usual trajectories, unless very specific conditions are met (see Chapter 5) – yet these are almost impossible to achieve in most SES, above all in those within developing countries and tropical regions. Thus, it is argued that there is a disjunction of the economic and conservation elements upon which the sustainable development paradigm is founded, enhanced by a power imbalance between the stronger unsustainable economic-development forces compared to the (weaker) conservation forces. With regard to the first part of the research question, here a *distinction between short- and long-term sustainability* needs to be done. Modelling results (Chapters 3-5) showed that several socio-economic and governance factors hinder short-term sustainability in coupled SESs under the current economic paradigm; namely monetary debt, the type of economic and production systems, technological development, and weak conservation forces (both top-down and bottom-up). Interestingly, alternative scenarios showed that these same factors could be also redirected to enhance short-term SES sustainability. These results align with Barnes (2006), who recognizes the benefits of capitalism by arguing that most economic elements under the current system (e.g. market, banks, credits, technology) will be necessary to create a sustainable, long-lasting society. Based on the dual role of the above-noted factors, it is argued that the current economic system is not inherently (i.e. by definition, *per se*) unsustainable; rather, the specific use of economic mechanisms and the behaviour of economic entities, as well as their decisions and relationships with regard to the environment, show a tendency to hinder short-term sustainable development under the current capitalist economic system.

With regard to the long-term sustainability – although this thesis rejects the idea that the system has to be replaced wholesale by a completely different one, Chapter 6 has shown the need for a gradual, most probably long-lasting, transition to achieve long-term sustainability. Thus, the above-noted short-term solutions may not be sufficient to enhance long-term sustainability. More specifically, there will be a need to re-adapt the current socio-economic system to be able to integrate, and fully account for, externalities and the value of natural capital. For this purpose, natural capital needs to be propertized and operationalised, for instance, by creating property rights for environmental commons that integrate an adequate long-term valuation and management of nature. While this goal is complex and challenging, a first step toward this direction may be to ascribe monetary value to ES through PES, as a mechanism that can start shifting capitalist forces driving economic growth to support long-term environmental conservation.

Chapter 7:
Epilogue

> "*Society is indeed a contract...between those who are living, those who are dead, and those who are to be born*"
>
> – Edmund Burke (Irish statesman, 1792, p. 359)

> "*The best way to predict the future…is to create it*"
>
> – Abraham Lincoln (16th President of the United States, 1809-1865)

7.1 How should the concept of sustainable development develop?

The concept of sustainable development (SD) officially emerged in 1987 from the report called 'Our Common Future', also known as the Brundtland Report (WCED, 1987). The report argued that boosting the economy, protecting natural resources, and ensuring social justice are not conflicting but interconnected and complementary goals. A healthy environment, the theory goes, provides the economy with essential natural resources. A thriving economy, in turn, allows society to invest in environmental protection and avoid injustices such as extreme poverty. And maintaining justice, by promoting freedom of opportunity and political participation, for example, ensures that natural resources are well managed and economic gains allocated fairly. Civilizations that have ignored these connections have suffered: consider the Easter Islanders, who depleted their forest resources and, thereby, entered into a spiral of economic difficulties that eventually led to their civilization's collapse (Diamond, 2005).

The development of the SD concept was based on merging previous well-known socio-economic and environment issues, under the umbrella of a new term. SD has been useful; above all, in terms of developing interdisciplinary debates that have made people more aware of the environmental problems our society faces nowadays (Beckerman, 1994). It has also enhanced holistic and interdisciplinary scientific approaches to answer different questions (Bawa and Seiler, 2009). Nevertheless, the concept of SD never gained full clarity, with abstract objectives that hindered the implementation of efficient policies and strategies towards achieving global sustainability (Victor, 2006). Its broadness and fuzzy nature has made it even more difficult to inform governments, corporations and NGOs on their role in building a more sustainable society (Victor, 2006). Moreover, the SD concept could have followed a diplomatic process based on devoting too much effort to lengthening the international community's wish list, instead of articulating and ranking the types of practical measures of serious policy-making. As an example, Agenda 21 embraced every goal offered up in anticipation of the Rio summit, but it set no specific priorities or targets, making it impossible to mobilize support for any strategy or to measure progress (UN, 1993).

As a consequence, SD, the compass that was designed to guide the way to just and viable economy and society, now points in all directions. The concept has now become fuzzier than ever before since it stresses the interconnection of ‘everything’. Hence, there is a need to return to Brundtland's fundamentals, where SD is viewed as a ‘fresh’ and practical framework for every aspect of governance. In this regard, the recently published Sustainable Development Goals (SDGs) (ICSU, ISSC, 2015) offer an opportunity to revive the concept. The SDGs cover all sectors of society through a novel approach to global governance, based on a goal-setting strategy (Bierman *et al.*, 2017). This policy-oriented approach could help reducing the vulnerability of the SD term regarding special interest groups (which may distort the concept for their own benefit) and reduce its ‘fuzziness’. However, if the SD concept is to be revived and the SDGs are to be implemented through effective policies and strategies, various challenges need to be addressed. First, states should show a formalized commitment to the SDGs (Bierman *et al.*, 2017), where responses and plans at local, national and global levels are coordinated (Bowen *et al.*, 2017; Kanie and Bierman, 2017). A second challenge lies on avoiding the negative consequences from responses to goals in isolation (Gao and Bryan, 2017; Nilsson, 2017; Nilsson *et al.*, 2016), for which accessing information and resources to understand the goals and how to respond is an essential aspect (ICSU, 2016; United Nations, 2016). In this regard, it is important to address trade-offs among SDGs, such as the need to increase food security and production, while enhancing conservation of ecosystems and the environment in general. Thus, integrative and holistic approaches to explore synergies among SDGs should be considered.

Furthermore, Victor (2006) suggests different courses of action in order to revive the concept of SD and to move towards scenarios that could help us achieve the SDGs. One of them states that there is a need to drop the environmental bias that has hijacked the entire SD movement over the past few decades. Victor (2006) argues that the SD agenda has been ‘dominated’ by the environmental dimension, where well-organized institutions (e.g. NGOs) have managed to make the concept their own flag. Although the SD-environment link may be positive for conservation purposes, it can also lead to misinterpretation of environmental issues, or even the rejection or lack of interest on the SD concept by institutions and actors from outside the environmental fields. Interestingly, the action of moving environmental issues to the top of SD agenda has not lead to more interdisciplinary debates within the environmental pillar. In fact, research by Schoolman *et al*. (2010) shows that scientific papers on the environmental dimension of sustainability are less integrative in terms of drawing on material outside the discipline, compared to papers from the economic and social dimensions. This demonstrates that research performed under the environmental ‘pillar’ needs to move towards more interdisciplinary approaches. Similarly, the SD concept has become an attractive term for some other interest groups (Kates *et al*., 2005), such as economists. Thus, the latter would be frequently employing the term as a vague gesture to the need for environmental conservation in the context of prioritizing economic growth (Wu, 2013).

Another factor that could help the SDGs becoming more relevant at governance and policy-making levels has to do with scaling. The ecosystems and economies of nations are interdependent, and the problems they face require global solutions. However, because the SDGs consist of a broad set of global principles, countries find it difficult to achieve, track or evaluate any of the SDGs. Thus, there is a need to favour local and regional decisions over global ambitions (Bierman *et al*., 2017). Accommodating local preferences and capabilities within the SDGs could not only make the SD concept to gain practical relevance, but also help reducing its ‘fuzziness’ and vulnerability. The problem of keeping the concept (solely) at global and/or theoretical levels can be explained through an example: the authors of the Kyoto Protocol envisioned a single global trading system with a single global price for carbon (UN, 1998). However, such a uniform system is not being implemented because the institutions that allocate credits, monitor compliance, and enforce agreements operate mainly at the local and national levels (Cramton *et al*., 2017). As a result, emissions-trading systems are emerging from the bottom up.

This more pragmatic, local approach may be more demanding for governments, even less ideologically satisfying. However, it is a necessary action if SDGs are to be applied through efficient policies at local and regional scales (Victor, 2006). Furthermore, in order for global ambitions to be translated into national and lower contexts, processes that monitor, evaluate and assess progress with regards to SDGs need to be developed (Bhaduri *et al*., 2016; Haski-Leventhal, 2015). In this regard, trade-off approaches and calculations may be used and considered as valid methods to downscale the SDGs. For instance, the models presented in this thesis (Chapters 3-5) address various different SDGs (see Chapter 2) by modelling trade-offs and synergies among different ES and biodiversity at regional and sub-national scales. Prior to developing these models, the fundamentals of SD were studied, as well as their applicability to each case-study by considering their own particular socio-economic, cultural, political and environmental contexts. The global nature of the SDGs was thereby downscaled to lower levels through specific empirical models, thus presenting a balance between the theory (global) and empirical application (local).

Back in 1987, the concept of SD was a smart and attractive idea because nobody really knew what it meant. Nowadays, the power and relevance of the concept has decreased, while there are different ways to understand the term 'sustainability'. For some actors, this idea is an obstacle to economic growth and the market economy. For some others, it is a vague concept that gives opportunities for people with different agenda to interpret it to suit their own interests. Therefore, fixing the concept of SD will require going back to its origins in 1987, and especially stressing the integration of economic and ecological systems in order to create long-lasting viable systems. Because local needs and interests will necessarily vary, SD must be redefined repeatedly, from the bottom up, wherever it is to be put into practice. Competent local institutions can then decide how to set and pursue their own priorities – under the umbrella of a broad and global set of priorities such as the SDGs. SD may thereby recover its relevance and appeal again as a non-negotiable and necessary goal for the good of this planet and our society.

7.2 Final reflections

This thesis showed that, by standing back far enough from a problem, the problem comes into focus, and it becomes clearer which interrelationship of factors is responsible for the patterns of behaviour and system outcomes. If we are to achieve sustainability, the economy must be viewed in its proper perspective as a subsystem of the larger, and more important, environmental system of which it is a part. And this is the main principle of the social-ecological systems science performed in this thesis: humans must be seen as a *part of*, not apart from, nature (Berkes and Folke, 1998).

Unsustainable development is the most persistent, structural and dramatic problem facing the society and global economy today. On the one hand, representing owners of capital, are powerful profit-maximizing corporations, which dominate the economy. On the other hand, representing future generations, nonhuman species, and millions of humans with unmet needs, are, almost nothing. The inner nature of corporations makes them diminish common wealth and natural capital, while their only obvious counterweight is government – yet government is significantly influenced by these same corporations.

The reason why capitalism prevents and distorts a just, democratic and sustainable world-system is simple. Democracy is an open system, and economic power can easily infect it. Capitalism is a gated system; its bastions are not easily accessed. Capital's primacy thus is not an accident, nor the fault of any corporation. It is what happens when capitalism inhabits democracy. As a result, capitalism has completely *disconnected* the socio-economic system from the natural system, and we are in need to *reconnect* them again. The current version of capitalism cannot, therefore, last for much longer. The new version has to adequately value and account for the natural capital in which our long-term wellbeing depends. Just as we receive the natural capital as a shared gift, so we have the duty to pass it on in at least the same condition as we received it. If we can add to its value, so much the better, but at a minimum we must not degrade it, and we certainly have no right to destroy it.

Going back to the quote from Edmund Burke (1792) shown at the beginning of this chapter: "*Society is indeed a contract...between those who are living, those who are dead, and those who are to be born*". Major historical events illustrate successful examples of structural changes that took place in our society to provide continuous intergenerational benefits. One example is the Social Security. It was imagined, designed, and installed early in the twentieth century in response to what was then an emerging crisis: the impoverishment of millions too old to work (SSA-USA, 2018). The basic intergenerational contract was, and remains, simple: active workers collectively support retired workers, and in return the former are supported, in old age, by the next generation of workers.

We need a similar contract for sustainability. One that fixes the disregard that capitalism has for nature and future generations. For this purpose, we will need models that provide solutions with regards to achieving win-win economic-environmental scenarios. It will be a challenging trip. Yet, does this mean there is no hope? The window of opportunity is small, but not non-existent. This may be, perhaps, a thirty- to fifty-year project to bring the new capitalism into harmony with nature. And, most probably, the new 'capitalism' will involve a Faustian deal of some sort: if we want the goods, we must accept the bads. But, if we must make a deal with the Devil, I believe we can make a much better one than we presently have. I am confident that, if we understand how to get a better deal, we will get one. After all, our children and lots of other creatures are counting on us.

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Appendix A – Research outputs: publications, conferences and seminars

Scientific articles (submitted and under review)

- **Title**:
  Exploring the relationship between debt and environmental sustainability: An Agent-Based Modelling Approach.

  Status:
  Submitted to the journal *PLOS ONE* (Under Review).

  Authorship statement:
  Authors: Julen Gonzalez-Redin, J. Gary Polhill, Terence P. Dawson, Rosemary Hill, Iain J. Gordon

- **Title**:
  Sustainable development pathways are achievable in debt-based economies: A case-study in Indonesia

  Status:
  Submitted to the journal *Environmental Science & Policy* (Submitted).

  Authorship statement:
  Authors: Julen Gonzalez-Redin, J. Gary Polhill, Terence P. Dawson, Rosemary Hill, Iain J. Gordon

- **Title**:
  Exploring future sustainable landscapes in the Wet Tropics: An Agent-Based Modelling approach.

  Status:
  Submitted to the journal *Global Environmental Change* (Submitted).

  Authorship statement:
  Authors: Julen Gonzalez-Redin, Iain J. Gordon, Rosemary Hill, J. Gary Polhill, Terence P. Dawson.

Scientific articles (published)

Rognvald, S., Barton, D.N., Dick, J., Haines-Young, R., Madsen, A., Rusch, G., Termansen, M., Woods, H., Carvalho, L., Giuca, R.C., Luque, S., Odee, D., Saarikoski, H., Rusch, V., Vădineanu, A., Dunford, R., **Gonzalez-Redin, J.**, Verweij, P., Vikström, S., Adamescu, M. and Stange, E. (2017). Operationalizing ecosystem service assessment in Bayesian Belief Networks: Experiences within the OpenNESS project. *Ecosystem Services*, 29(C), pp. 452–464.

Gonzalez-Redin, J., Luque, S., Poggio, L., Smith, R. and Gimona, A. (2016). Spatial Bayesian Belief Networks as a planning decision tool for mapping ecosystem services trade-offs on forested landscapes. *Environmental Research*, 144, pp. 15–26.

Gonzalez-Redin, J. (2017). Drivers of unsustainability in tropical regions: Is there space for hope? *Protected*, 14, pp. 7.

Book chapters

Luque, S., **Gonzalez-Redin, J.** and Fürst, C. (2017). Mapping forest ecosystem services. In: Burkhard, B. and Maes, J. (Eds.). Ecosystem Services Mapping. Sofia: Pensoft Publishers pages, pp. 324–328.

Conference proceedings and presentations

Gonzalez-Redin, J., Polhill, J.G., Dawson, T.P., Hill, R. and Gordon, I.J. (2017). Sustainable development pathways in debt-based economies: An Agent-Based Modelling approach to palm oil production scenarios in Indonesia. In: *Social Simulation Conference (SSC)*. Dublin, Ireland, Sept. 26–29.

Gonzalez-Redin, J., Gordon, I.J., Hill, R., Polhill, J.G. and Dawson, T.P. (2017). Sustainable Futures in Tropical Landscapes: An Integrated Agent-based Modelling approach. In: *12th Conference of the European Society of Ecological Economics (ESSE)*. Budapest, Hungary, June 20–23.

Gonzalez-Redin, J., Polhill, J.G., Dawson, T.P., Gordon, I.J. and Hill, R. (2016). Examining the impact of debt-based economic systems on natural resources: An Agent-based modelling approach. In: *Social Simulation Conference (SSC)*. Rome, Italy, Sept. 19–23.

Gonzalez-Redin, J., Polhill, J.G., Dawson, T.P. and Gordon, I.J. (2015). Decoupling energy and natural resource use from economic growth: An Agent-Based Modelling approach. In: *21st International Congress on Modelling and Simulation (MODSIM).* Gold Coast, Queensland, Australia, Sept. 29 – 4 Dec.

Seminar presentations

Gonzalez-Redin, J. (2016). *Seminar presentation* at the Center for Tropical Environmental & Sustainability Science (TESS), Cairns, Queensland, Australia.

Gonzalez-Redin, J. (2016). *Seminar presentation* at the Social, Economic and Geographical Sciences (SEGS) Group, The James Hutton Institute, Aberdeen Scotland UK.

Appendix B – Overview, Design Concepts, and Details (ODD) protocols

See separate bound booklet